



**ESTIMATING ANCILLARY BENEFITS
OF CLIMATE POLICY
USING ECONOMY-WIDE MODELS:
THEORY AND APPLICATION
IN DEVELOPING COUNTRIES**

by

*David O'Connor**

**OECD Development Centre
Paris, France**

david.oconnor@oecd.org

www.oecd.org/dev

**Presented at the
EEPSEA Biennial Researchers' Workshop
Hanoi, 21-22 November 2001**

* The research on which this paper is based has been a collaborative effort. The author would like to thank his collaborators on various country studies: Maurizio Bussolo (India); Sébastien Dessus (Chile); Kristin Aunan, Terje Berntsen, Zhai Fan, and Haakon Vennemo (China). Naturally, any errors herein are the author's alone. Also, the views expressed are solely those of the author and should not be construed as those of the OECD, its Development Centre, or their respective Members.

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INTRODUCTION

Climate change may have unanticipated consequences; so too might climate policy. Suppose that government levies a tax on the carbon content of fuel. Doing so is intended to alter levels and patterns of energy use and thereby reduce greenhouse gas (GHG) emissions. At the same time, however, the tax will alter levels of other emissions – e.g., particulate matter, SO₂, NO_x, volatile organic compounds (VOCs), carbon monoxide (CO), and lead – all of which are associated with fossil fuel combustion. This in turn will affect local air quality and the health of the population. When analysts calculate the costs and benefits of climate policy, to a large extent they ignore any “ancillary benefits (costs)” in terms of changes in local pollution damages. This tends to bias any policy prescriptions flowing from such analyses towards a “wait-and-see” stance, since abatement costs (the costs of reducing CO₂ and other GHG emissions) are incurred almost immediately, while the benefits of climate change averted are not only distant in time but highly uncertain. Meanwhile, policy makers have many pressing near-term development (and even environmental) priorities to which to attend. Yet the impacts of climate policy on local pollution are real and, in principle at least, measurable. Recently, considerable attention has been devoted by the climate economics community to trying to arrive at plausible estimates of the magnitude of such ancillary benefits (cf. papers in OECD 2000b), in order to compare them with abatement costs and determine what scope there may be for “no regrets” GHG control measures.

This paper describes work that the OECD Development Centre has undertaken in the past two years to devise and implement a methodology for ancillary benefits estimation in developing countries, using economy-wide models. Section 1 presents a simple analytical framework for integrating ancillary benefits into climate economics. Section 2 then motivates the use of so-called “top-down” models for the analysis, weighing their advantages and disadvantages *vis-à-vis* “bottom-up” engineering-based models. It also describes the basic structure of the CGE model. Section 3 describes the other modules needed for the analysis, including the air dispersion model and the dose-response functions. Section 4 addresses questions of valuation of ancillary health effects, including the issues raised by benefits transfer across countries. Section 5 presents the welfare analysis resulting from the climate policy experiment and sensitivity analysis for India. These results are compared with others in the literature. Section 6 summarises and points in a few directions for future research.

1. SIMPLE ANALYTICS OF CARBON ABATEMENT COSTS AND ANCILLARY BENEFITS

In the simplest terms, climate policy can be thought of as any measure or set of measures designed to constrain an economy’s net GHG emissions below some baseline. For developed country Parties to the Kyoto Protocol, that baseline is usually 1990 emissions. For a developing country, it might be a business-as-usual (*BAU*) growth baseline. (*Note:* for simplicity, the discussion here is in terms of CO₂ emissions, but it could be extended to a multi-gas assessment; see OECD 2000a for one example.)

Figure 1.A presents a stylised picture of how total costs vary with abatement, suggesting that they increase at an increasing rate – i.e., the *marginal abatement cost* curve is convex to the origin. The figure also depicts a stylised ancillary benefits curve, which is shown as a ray from the origin (by definition, with zero abatement there are zero ancillary benefits), with the constant slope assuming – as a first approximation – the linearity of underlying dose-response functions and absence of any minimum exposure threshold. The epidemiological literature on mortality and morbidity effects of particulate exposure is broadly consistent with these assumptions; in any case, both the *ex ante* and *ex post* exposure levels in the cities of the developing countries studied here (Chile, India) are well above any possible threshold. The figure and the analysis abstract from the primary benefits of climate change averted, not because they are considered insignificant¹, but because they are judged to be too uncertain and distant in time to influence substantially policy making in most developing countries. The inclusion of primary benefits would complicate the analysis (bringing to the fore the question of choice of discount rate, since primary benefits occur some way in the future) but would not fundamentally alter the framework. (The addition of primary benefits in Figure 1.A. would shift the benefits curve up for a given abatement rate but probably only infinitesimally for any single country.)

Figure 1.1.A: Gross and Net Costs of CO₂ Abatement

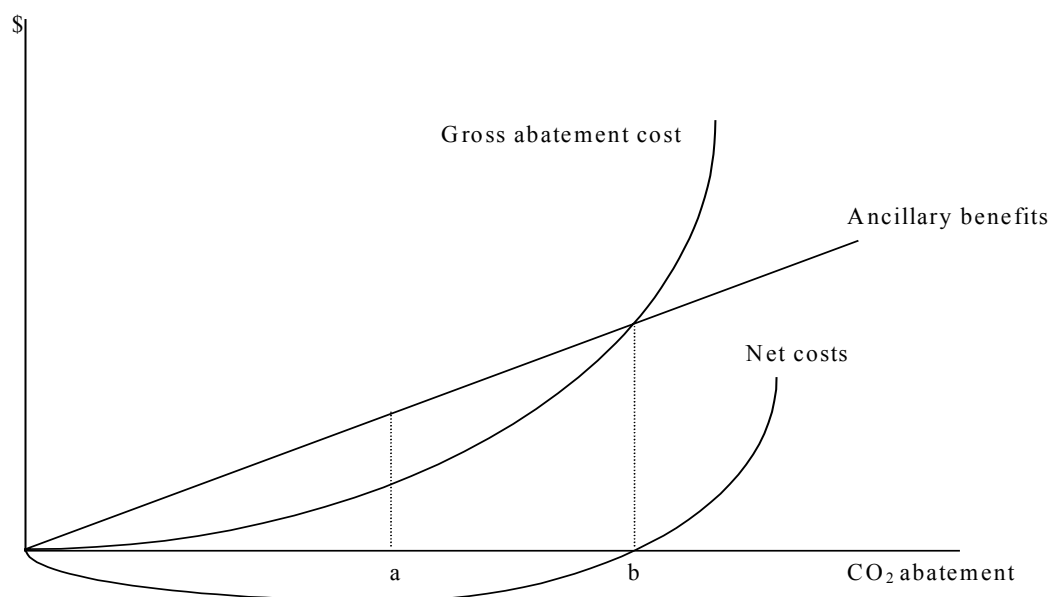
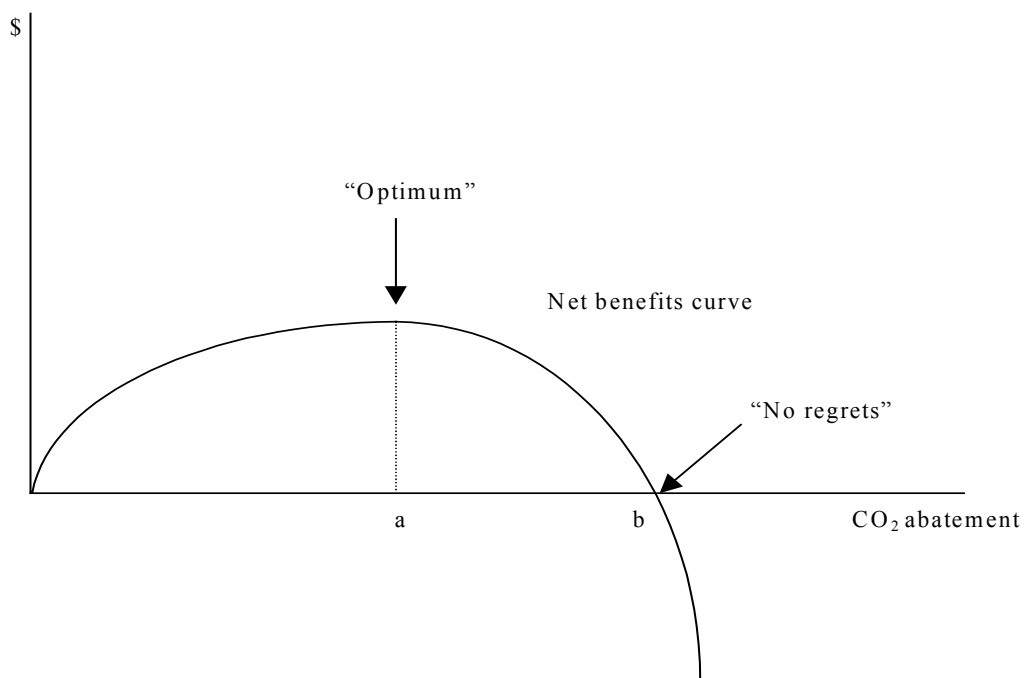


Figure 1.1.B: “Optimal” and “No Regrets” CO₂ Abatement



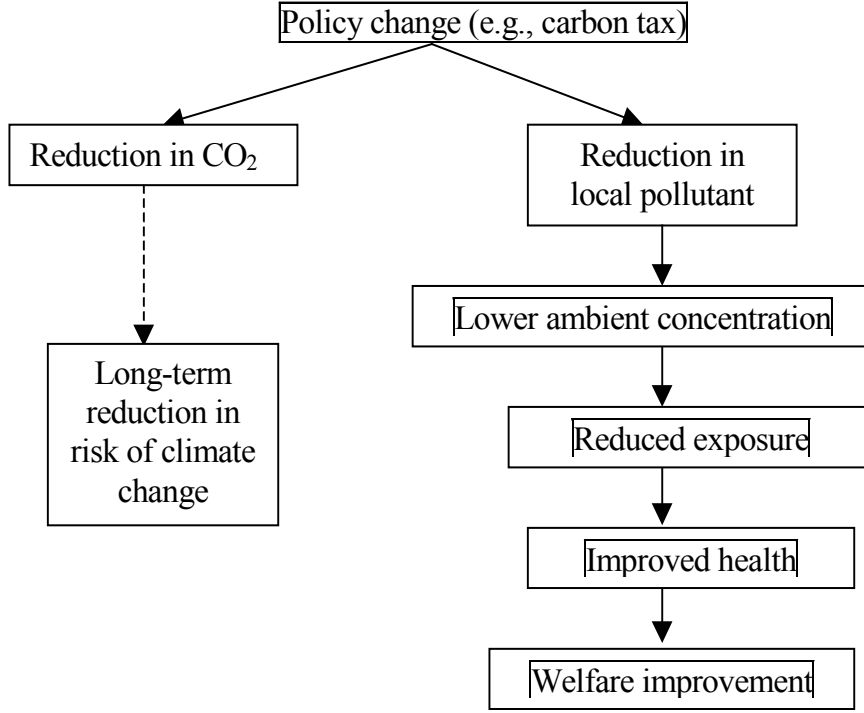
Through inversion of the net cost curve in Figure 1.1.A, we get the net benefits curve in Figure 1.1.B; as drawn, these are positive over some range, peaking at abatement rate a

before declining towards zero at point b (the so-called “no regrets” threshold) before turning negative. An “optimal” climate policy would seek to maximise the net benefits (again bearing in mind the absence from consideration of primary climate benefits), so “optimal” abatement would be lower than the maximum “no regrets” rate.

The costs depicted in Figure 1.1 are those of limiting an economy’s emissions of CO₂, which can be done through one or more of the following: (a) reducing the overall level of economic activity; (b) reducing the energy-intensity of a given set of activities; (c) fuel switching from high-carbon to low-carbon fuel or carbon-free energy sources for a given level of energy use; (d) reallocating resources away from energy-intensive sectors. If the economy is operating efficiently in an initial equilibrium, any one of these actions involves an opportunity cost. It is only if one assumes pre-existing inefficiencies – e.g., in energy input use – that the gross abatement cost curve would dip below the x -axis over an initial range of abatement. In this event, the net cost curve also shifts down proportionally and the “no regrets” abatement range is further increased.

The ancillary benefits curve is a construct involving several intermediate steps between the policy shock and the change in welfare, as measured by equivalent variation (the functional specification of the welfare change is given below). These steps are depicted in Figure 1.2. The crucial link in the chain is from the climate policy (say, a carbon tax) to the impact on other pollutants. Taking particulates for purposes of illustration, we need to know how a carbon tax – levied for example on the carbon content of fuel – translates into reductions in particulate emissions, in other words, the cross price elasticity of particulates with respect to carbon (ξ_{pc}). The higher is ξ_{pc} , the greater will be the effect on particulate emissions of a given carbon tax. What determines the value of ξ_{pc} ? Most importantly, it depends on the extent to which the two pollutants have been “de-linked” through prior particulate pollution controls – i.e., the enforcement of particulate standards and the resultant behavioural changes of polluters, e.g., through installation of end-of-stack capture technologies.

Figure 1.2: Links in chain from policy measure to welfare change



It is generally the case that the high-income OECD countries have gone farther than developing countries in controlling local pollution emissions, with the result that ξ_{pc} values are likely on average to be lower in the former than in the latter. If so, this suggests a hypothesis about the relationship between a given carbon tax and the size of expected ancillary benefits, viz., that *the lower a country's per capita income, ceteris paribus, the larger are the effects of a carbon tax on local pollution, hence the bigger the ancillary benefits (measured as changes in health endpoints per tonne of carbon reduction)*. Formally,

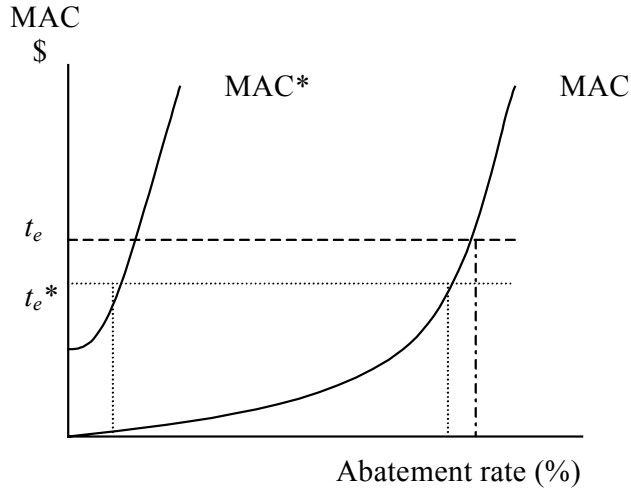
$$(rgdp)_i < (rgdp)_j \quad \Rightarrow \quad (\xi_{pc})_i > (\xi_{pc})_j \quad \Rightarrow \quad (AB_i | t_c) > (AB_j | t_c)$$

where $rgdp_{i,j}$ refers to real *per capita* GDP of any two countries i and j , t_c is the carbon tax rate and $AB_{i,j}$ are the ancillary benefits for countries i and j (measured in physical units – e.g., premature deaths avoided per tonne carbon reduction). Whether this translates into larger monetised welfare gains depends on the relative incomes of the two countries and, by implication, their willingness to pay (*WTP*) for the expected health improvements.

Figure 1.3 presents this analysis in graphical terms, showing the marginal abatement cost curves for local pollution for a low-income country (MAC) and a high-income country (MAC*). The latter is shifted to the left because of the prior abatement of local pollution, so the response to a carbon tax already finds the high-income country on the steeply

ascending portion of its marginal abatement cost curve. Also in the high-income country, because of the relatively low cross price elasticity of carbon and local pollution, a given carbon tax translates into a lower effective tax on the latter – t_e^* versus t_e . The combination of these two effects implies a lower *post-tax* equilibrium level of local pollution abatement, hence, lower ancillary benefits in the high-income country than in the low-income one.

**Figure 1.3: Marginal Abatement Costs and Rates
for Local Pollutants, Developed and Developing Countries**



One possible complication is that the imposition of a carbon tax may not be the most efficient way to achieve a given reduction in local pollution. In other words, while the MAC curve traces out the path of incremental costs assuming that lower cost abatement options are always chosen before higher cost ones, some of those options may not become relevant in the face of a tax on carbon, viz., those options that lower emissions of the local pollutant without affecting carbon emissions (or perhaps even increasing carbon emissions). The question of how important these options are likely to be is ultimately an empirical one. End-of-pipe particulate capture technologies are one example: not only do they not reduce carbon emissions, but the fuel used to run the equipment may actually raise those emissions somewhat. A few studies have sought to examine the degree of correlation between cost-effective local pollution control technologies and cost-effective carbon abatement ones.

Eskeland and Xie (1997) compare various abatement technologies for mobile source air pollution in Mexico City, in terms of cost effectiveness in reducing a weighted local toxicity index *versus* reducing GHG emissions. They find that, excluding shifts in transport mode and demand management measures (e.g., a pollution tax on motor fuels), the rank correlation between local cost-effectiveness and global cost-effectiveness is rather weak. Out of some 26 identified control measures, stricter motor vehicle emission standards are the ones exhibiting the highest correlation in the two sorts of cost-effectiveness, largely because these standards would improve the fuel efficiency of gasoline-powered vehicles. Whether imposition of an environmental fuel tax would have to be part of a cost-effective strategy for local pollution control depends critically on the

own-price elasticity of demand for polluting fuels. In another study for Mexico, Eskeland and Feyzioğlu (1997) find that both in the short term and in the medium term demand for gasoline is fairly price-elastic, suggesting that a pollution-related gasoline tax would yield a rather strong behavioural response and would thus be a cost-effective policy instrument for realising local air quality improvements. A review of demand elasticity estimates for gasoline (Dahl 1995) supports the result that a tax could be a potent environmental policy instrument.

Cifuentes *et al.* perform a similar exercise for Santiago, Chile, but show a much stronger association between cost-effectiveness in the two dimensions (reduction in local pollution, as measured by $PM_{2.5}$, and reduction in carbon emissions) (see EPA 2000). In a diagram showing rank order of cost-effectiveness of different technical options along the two axes, a large proportion of such options (which unlike in Eskeland and Xie are not limited to transport) cluster along the 45-degree line, suggesting that those ranking high in $PM_{2.5}$ cost-effectiveness do likewise in carbon cost-effectiveness. Of particular interest is the price sensitivity of some technical options, with the conversion of buses to compressed natural gas (CNG) looking very promising in terms of both types of cost-effectiveness at 1999 prices, but far less attractive in terms of $PM_{2.5}$ abatement cost-effectiveness at the higher 2000 gas prices.

Lvovsky *et al.* (1999) take a slightly different approach, comparing different control strategies in terms of their local and global environmental benefits. They find – given the nature of the model relating particulate emissions from different sources to ambient concentrations and population exposure (of which, see further discussion in the next chapter) – that those measures that have the largest local benefits in terms of reduced health impacts (e.g., control of emissions from small stoves and boilers and of diesel motor vehicle emissions) do not yield the largest global benefits (in terms of GHG reductions), for which electricity fuel switching is far more important.

2. THE ECONOMIC MODEL

The analysis of climate policy is inherently interdisciplinary. In this paper, we are concerned primarily with the *economics* of climate policy, but elements of other disciplines like engineering, atmospheric sciences, health sciences, and agronomy inevitably enter into the picture.

2.1. TOP-DOWN VERSUS BOTTOM-UP MODELLING APPROACHES

There are two basic modelling approaches to estimating the costs and benefits of climate policy – “bottom-up” engineering models and “top-down” economy-wide models. The former tend to be based on least-cost technical options while the latter focus more on behavioural responses to price and income changes.

In principle, one would expect the two approaches to yield broadly identical results, since they refer to the same set of economic agents and technologies. In practice, however, “bottom-up” models tend to yield lower abatement cost estimates than “top-down” ones.

Kolstad and Toman (2001) offer a plausible explanation along the following lines. Beginning with a given policy shock (say, a gasoline tax), the former approach would list all the technically feasible ways of reducing gasoline consumption in response to the shock, including seemingly costless responses like trip consolidation. The latter approach begins from observation of actual past behavioural responses to gasoline price changes to estimate what the reaction would be to this new policy shock. That response depends on how vehicle use enters into consumers' utility functions, e.g., how they value time and convenience. Advocates of a "bottom-up" approach would argue that past behaviour is not necessarily a reliable guide to future behaviour, that if people were educated about how they could save energy at little or no cost they would choose to do so. The "top-down" approach essentially takes the current state of knowledge as a given, implicitly assuming that information about cost-saving opportunities is complete.

One might argue that the "bottom-up" approach is patently more realistic in allowing for changes in people's state of knowledge. The main limitation of this approach, however, is that it is not able to capture the full complexity of economic interactions in a consistent framework. This is the strength of "top-down" models, notably computable general equilibrium (CGE) models. They become especially useful when one would like to simulate the effects of something like a carbon tax or an emissions cap, since these will have economy-wide effects. To be able to compare economy-wide costs with benefits (ancillary or otherwise), one needs to be able to see the big picture, to consider not only technical possibilities but behavioural relationships, not only sector-specific abatement options but cross-sectoral shifts in resource allocation.

2.2. MODEL STRUCTURE

Our CGE model structure derives originally from the OECD's GREEN model (see Burniaux *et al.*, 1992), which is a multi-region global model for simulating climate policy. GREEN has been adapted for use at the Development Centre, with a number of progeny in the form of country-specific models for environment-economy analysis (cf. Beghin, Dessus, Roland-Holst and van der Mensbrugghe, 1996). The basic technical coefficients in the various country models used for our analysis (Chile, China, India) are derived from social accounting matrices (SAMs) built, *inter alia*, from input-output tables, national accounts, industry surveys, and household expenditure surveys. These matrices show the flows of income and expenditure in the economy in the base year, including intermediate input purchases by each sector. Figure 2.1 presents a simplified SAM, with sector of origin in columns and sector of destination in rows.

Figure 2.1: Simple SAM Structure and Accounting Relationships

	Suppl. 1	Househ. 2	Gov. 3	CapAcc. 4	ROW 5	
1 Suppliers	IC	C	G	I	E	Demand
2 Households	Y					Income
3 Government		T				Receipts
4 Capital Account		Sh	Sg		Sf	Savings
5 Rest Of the World	M					Imports
	Supply	Expend	Expend	Investment	ForExchg	

- 1) $Y + M = C + G + I + E$ *GNP: Value Added + Imports*
 $= Consumption + Gov Expenditure + Investment + Exports$
- 2) $C + T + Sh = Y$ *Domestic Income: C + Direct Taxes + Hh Savings = Income*
- 3) $G + Sg = T$ *Government budget: G + Gov. Savings = T*
- 4) $I = Sh + Sg + Sf$ *Investment - Saving: I = Sh + Sg + Foreign Savings*
- 5) $E + Sf = M$ *Foreign Balance*

The behaviour of economic agents is modelled according to neo-classical assumptions of utility- and profit-maximisation. All markets clear at equilibrium prices. The economy consists of multiple production sectors but there is only a single representative household (this last assumption can of course be relaxed with sufficient micro data on household expenditure classes).

Production and Capital Accumulation

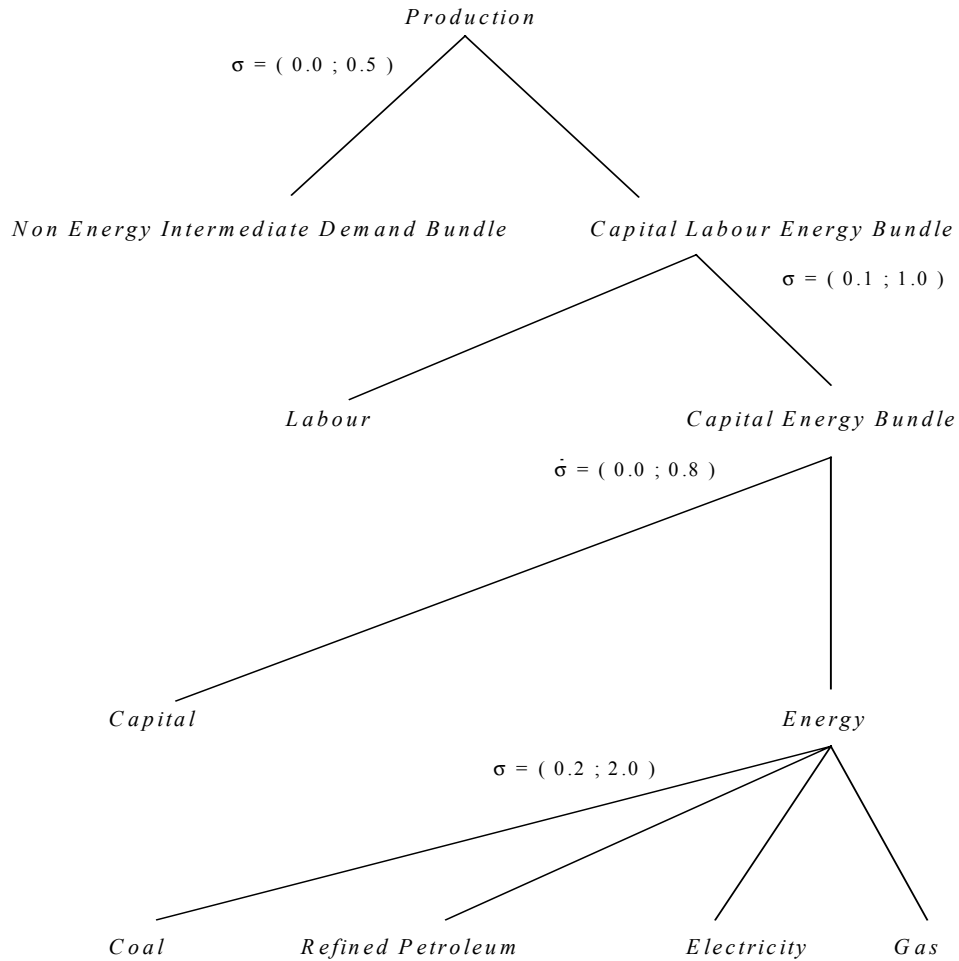
The model structure is dynamic recursive: *dynamic* in that a change in current period savings volume influences capital accumulation in the following period; *recursive* in that agents are assumed to be myopic, basing their consumption and investment decisions on current prices and quantities rather than expectations about future ones. Exogenous growth rates are assumed for various other factors that affect the growth path of the economy, such as population and labour supply, labour and capital productivity, and energy efficiency (see discussion below).

Capital is putty/semi-putty, with the installed base enjoying lower substitution possibilities than new investment. In each sector, production is undertaken according to a nested constant elasticity of substitution (CES), constant returns to scale (CRS) production function; the nesting is shown schematically in Figure 2.2. The old vintage and new vintage substitution elasticities are given in parentheses at each branch point. For example, within the energy nest, existing fuels substitute for one another with an elasticity of 0.2 while with new energy investment the inter-fuel substitution elasticity rises to 2.0. [Note that intermediates have zero substitution possibilities among themselves; this is because of the fixed (Leontief) coefficients of the I/O tables. A more sophisticated analysis would relax this assumption, e.g., by estimating econometrically the substitution elasticities among intermediate inputs in response to relative price changes, using historical I/O tables. McKibbin and Wilcoxon (1995) have done this for the United States using I/O tables extending from 1958 to 1982.]

These substitution elasticities are crucial to the determination of abatement costs (in the broad sense of total welfare losses following a policy shock, before accounting for

external benefits). The higher are substitution elasticities, *ceteris paribus*, the lower will be the costs of adjustment to the new policy equilibrium. Since those elasticities differ markedly between old vintage and new vintage capital, this implies that the costs will be lower the higher the investment rate, hence, the faster the turnover of the capital stock. This is the main reason why most global models show GHG abatement costs to be lower in major developing countries than in developed countries.

Figure 2.2: Production Nesting



Notes:

1. Each nest represents a different CES bundle. Substitution elasticities separated by a semi-colon indicate, respectively, the central CES substitution elasticity for *old* capital and for *new* capital. The elasticity may take the value zero. Because of the putty/semi-putty specification, the nesting is replicated for each type of capital, i.e. *old* and *new*. The values of the substitution elasticity will generally differ depending on the capital vintage, with typically lower elasticities for *old* capital.
2. Intermediate demand, both energy and non-energy, is further decomposed by region of origin according to the Armington specification. However, the Armington function is specified at the border and is not industry specific.

An important issue in a developing country context is how a given climate policy is likely to affect the substitution between traditional fuels (e.g. fuelwood, crop residue, animal dung, other biomass) and various commercial energy sources. For instance, if the main commercial cooking fuel in rural areas is either natural gas or kerosene, and if a carbon

tax should raise its relative price, poor households especially may prefer to revert to using biomass fuels. In that case, there could well be an adverse health effect on a sizeable segment of the population from increased indoor air pollution (cf. Smith 1993). As Shukla (1996) observes, this sort of inter-fuel substitution is normally overlooked in the models developed for OECD countries. The author is not aware of any credible estimates of substitution elasticities between these non-commercial fuels and commercial fuels in developing countries. Ideally, one would want to model the non-commercial fuel sector, but the absence of a separate biomass fuel sector² from the underlying I/O tables makes this infeasible for present purposes.

Income Distribution and Household Utility

Labour income is allocated to the representative household. Likewise capital revenues are distributed among households, corporations and the rest of the world. Corporations save and invest the after-tax residual of that revenue. Private consumption demand is obtained through maximisation of a household utility function following the Extended Linear Expenditure System (Lluch, 1973). Household utility, a function of consumption of different goods and saving, is not influenced by environment quality. The ELES specification of utility avoids the limitations of CGE models that assume Cobb-Douglas or Constant Elasticity of Substitution (CES) utility. The latter two imply unitary income elasticity of demand, thus failing to account for the way changes in income can affect the structural adjustment of the economy to exogenous shocks. Income elasticities differ by product, varying in a range from 0.50 for basic agricultural products to 1.30 for services. Thus, following Engel's law, with rising disposable income a progressively smaller fraction gets spent on food.

International Trade

The model assumes imperfect substitution among goods originating from different geographical areas (Armington 1969). Import demand results from a CES aggregation function of domestic and imported goods. Export supply is symmetrically modelled as a Constant Elasticity of Transformation (CET) function. Producers decide to allocate their output to domestic or foreign markets responding to relative prices. Elasticities between domestic and foreign products are of comparable magnitude for import demand and export supply. Their values are 3.00 for agricultural goods, 2.00 for manufactured goods and 1.50 for services. The small country assumption holds, Chile being unable to change world prices; thus, its imports and exports prices are exogenous. Capital transfers are exogenous as well. The balance of payments equilibrium therefore determines the final value for the current account.

Model Closure

The trade (or balance of payments) closure rule is given by:

$$S_f + \sum_i P^{FSAV} P_i^{World} X_i^E = \sum_i P^{FSAV} P_i^{World} X_i^M$$

On one side of the balance sheet are exports, evaluated at world prices, and net foreign saving. On the other side of the balance sheet are imports evaluated at world prices (excluding tariffs). Any price in the model can be chosen as the numéraire. In this case, the foreign saving price index, P^{SAVF} , has been designated as the numéraire, and its value is always set to 1. Note that if imports exceed exports, then net foreign savings are positive (representing net foreign borrowing) and equal to the difference.

Other closure conditions apply to the government budget deficit and total savings/investment balance. The government budget surplus/deficit is taken as exogenous and the household income tax schedule shifts in order to achieve the predetermined net government position. Total investment must equal the sum of savings originating from households, government and rest of the world.

Emissions

Emissions are principally determined by intermediate or final consumption of polluting inputs, mostly fossil fuels. In addition, certain industries have emissions not directly linked to input consumption but related instead to their output levels (e.g., fugitive emissions, as with natural gas leakage and volatile organic compounds). It is assumed that labour and capital do not pollute. Emission coefficients associated with each type of consumption and production are derived originally from the World Bank's IPPS project, which used toxic release inventory (TRI) data to establish sectoral pollution intensities for the United States (see Hettige *et al.*, 1995). Output-based emissions coefficients have one major drawback, viz., that – once fixed –they do not allow a given output to be produced with fewer emissions, e.g., by using a different input mix. The only way to reduce emissions is to reduce output. In reality, for most air pollutants, there are three main ways of lowering sectoral emissions: in addition to reducing output, one can alter the input mix, e.g., consuming fewer polluting inputs, or capture the pollutants at the end of the stack through an abatement technology. While few CGE models incorporate an abatement technology (for an exception see Garbaccio *et al.* 2000, who assume an exogenous rate of abatement technology advance), such models do allow for a variety of substitutions away from polluting inputs, e.g.: low-carbon fuels for high-carbon fuels; non-fossil fuel energy for fossil fuels; non-energy inputs for energy (e.g., installation of process control equipment); energy-conserving inputs for energy-using inputs; less energy-intensive goods for more energy-intensive ones (e.g., more durable and recyclable plastics); finally, imports of energy-intensive goods for domestic production of same (Jorgenson *et al.* 2000).

Dessus *et al.* (1994) transform these output-based intensities into input-based coefficients by regressing sectoral emissions on sectoral input use, with the unallocated portion of emissions (the residual) attributed to pure process emissions. Formally, the total amount of a given polluting emission takes the following form:

$$E = \sum_i \sum_j \alpha_j C_{i,j} + \sum_i \beta_i X P_i + \sum_j \alpha_j X A_j$$

where i is the sector index, j the consumed product index, C intermediate consumption, XP output, XA final consumption, α_j the emission volume associated with one unit consumption of product j and β_i the emission volume associated with one unit production of sector i . Thus, the first two elements of the right hand side expression represent production-related emissions, the third one consumption-related emissions.

There are six air pollutants considered in the analysis of climate policy and its ancillary benefits: carbon dioxide (CO₂) – the main greenhouse gas, total suspended particulates, sulphur dioxide (SO₂), nitrogen dioxide (NO₂), volatile organic compounds (VOCs), and carbon monoxide (CO).

Energy Efficiency Improvements

Whereas bottom up models very often find scope for “win-win” technical improvements – e.g., in energy efficiency – that both reduce costs and improve the environment, CGE models start from the assumption that market actors are already making optimal choices about whether and when to use specific technologies (Edmonds *et al.* 2000). If a technology is not being employed this is because it is not profitable to do so. That having been said, most CGE models have their own version of “win-win” in the form of energy-saving technical progress, or costless energy savings, reflected in an autonomous energy efficiency improvement (AEEI) factor. The AEEI rate is normally set to reflect recent historical experience and some judgment about its likely sustainability. One rationale for incorporating AEEI into the analysis is the recognition that technological improvements, even if initially stimulated by price changes, persist even after the stimulus has been removed or diminished. Thus, once many energy efficiency improvements were introduced in response to the 1970s oil shocks, they came to dominate earlier technologies. In principle, it would be possible to endogenise the rate of AEEI by making it vary with energy price changes. Ideally, one would like to be able fully to endogenise energy efficiency and other technological improvements, making them responsive not just to relative price changes but to R&D expenditures, induced in part of course by those changes but more generally by the expectation of higher profits³. To do so requires, however, much firmer empirical evidence than currently exists on the way various factors influence not just the rate and direction of innovation but its commercial adoption. The returns to investment in new technology are usually highly uncertain.

Policy Instruments

There are essentially two sorts of flexible instrument for climate policy, carbon taxes and quantitative restrictions with tradable permits. In choosing between the two, the degree of uncertainty facing policy makers regarding the abatement cost and benefits functions needs to be considered. Suppose for the moment that the positions of both the ancillary benefits curve and the abatement cost curve are uncertain. As Weitzman (1974) has shown, mis-estimation of the benefits curve will result in equivalent welfare losses whether a tax or a permit scheme is used. In the case of a mis-estimated abatement cost curve, however, which instrument is preferred depends on the relative slopes of the marginal benefits and marginal cost curves (see exposition in Baumol and Oates 1988). If

the cost curve is steeper than the benefits curve, a tax yields smaller welfare losses than a permit scheme. Indeed, this is the case here, with marginal ancillary benefits being almost constant in abatement (a function of the linearity of the underlying relationships), while abatement costs rise steeply beyond some modest level of abatement. Thus, we choose to implement the climate policy as a tax on carbon content of fuels sufficient to achieve a given CO₂ reduction relative to a growth baseline. The final year for the climate policy simulation is 2010, the mid-point of the first Kyoto Protocol control period. One could also consider a longer time horizon, say, to 2020. The effect would be to lower the costs of achieving any given CO₂ reduction target relative to 2010, since more capital stock could be turned over. At the same time, ancillary benefits of climate policy would also be reduced, since the AEEI would have another decade in which to reduce the energy intensity of GDP.

The effect of a carbon tax on the economy depends importantly on two factors: (i) prior tax-induced or other economic distortions, with which the carbon tax may interact; (ii) the way in which revenues from the tax are recycled. Where prior distortions are present, the carbon tax may either amplify them or mute them, in the latter case yielding a “double dividend”. In the absence of distortions, the tax ought to be designed so as not to introduce new ones. This would normally call for a lump-sum transfer of revenues back to households, e.g., via a reduction in direct taxes. This is indeed the basic recycling rule employed here.

Welfare

There are two sorts of welfare change that need to be evaluated in the current context: those resulting from changes in prices of goods and services resulting from the introduction of a carbon tax; those resulting from changes in expenditure required to maintain a given state of health. The latter, in turn, can be broken into morbidity-related expenditures and mortality-related expenditures. The former include primarily the costs of self-medication and institutional medical care, though they also include the opportunity cost of time lost to illness or restricted activity. These are collectively referred to as the “costs of illness” and a lower bound is given by the observable costs of treatment, which represent real expenditures on the services of the health care sector and the products of the pharmaceutical industry. The latter refer primarily to the willingness to pay for reduced risk of premature death. This *WTP* for reduced mortality risk does not correspond to any readily observable market transactions and, for this reason, we normally rely on indirect measures like contingent valuation surveys or hedonic wage studies.

Where costs-of-illness estimates exist, these welfare changes can be endogenised in the model by assuming that, with lower pollution levels, a smaller outlay is required on health care expenditures to maintain a given health status of the population. Those reduced health-care expenditures free up resources to be spent on other goods and services. Mortality benefits, on the other hand, are treated exogenously from the CGE modelling framework. This requires the imposition of separability conditions on individuals’ utility functions, implying that the utility of reduced mortality risk is

independent of the consumption levels of various commodities (Boyd *et al.* 1995). Also, it should be noted that changes in the health status of the population and in mortality are assumed not to have a significant effect on the supply of labour (this assumption could of course be relaxed).

The welfare change from a climate policy experiment consists of three parts, as shown in the following equation for period t :

$$\Delta W = (y^* - y) - (E(p^*, u^*) - E(p, u^*)) - (D^* - D)$$

where y is disposable income, p the price system, u is utility, E the health expenditure function, D the monetary value of the change in mortality (*Deaths*), and the star exponent denotes the *with-policy* state. The first term, $y^* - y$, is the conventional equivalent variation (*EV*) measure of welfare change, calculated endogenously by the model (see definition in Kolstad 2000, p.303) and equivalent to the household disposable income loss $-(y^* - y)$ – in the new equilibrium evaluated at the new set of relative prices. The second term represents the difference in health care expenditures required to achieve the improved *with policy* health status (the second term, with a negative sign). The third term represents the change in mortality induced by the policy, evaluated at the so-called “value of a statistical life” (VSL^4) (if premature mortality goes down as expected, this term is also negative). Whether the overall welfare change of a given policy is positive or negative depends on the relative magnitudes of the first term (the measured income losses from the carbon tax) and the next two terms combined (the ancillary benefits). Thus, the maximum “no regrets” abatement of CO_2 is given by:

$$(y^* - y) = (E(p^*, u^*) - E(p, u^*)) - (D^* - D) .$$

3. MODELLING DISPERSION AND CONCENTRATION, EXPOSURE AND DAMAGES

A CGE modelling framework poses no problem for the analysis of the costs and primary benefits of global climate policy, since GHGs are truly global externalities, with the location of emissions having no bearing on temperature rise and the greenhouse effect. Ancillary benefits are quite different, however, in that they tend to rather localised. As a result, spatial detail of the models matters much more. Most CGE models, however, are not designed for analysis below the level of the national economy. This limits their usefulness in ancillary benefits analysis. The emissions generated by the economic model have no spatial markers (they are simply “national” emissions), so how does one know the respective contributions of different sources to air pollution in a specific city or other area of concern? Geographic location of emissions, stack heights of emitting sources, local temperature and meteorological conditions, population distribution and location of valuable assets vulnerable to pollution damage all matter to the nature and size of impacts⁵. In an ideal world, all these data would be available and one could employ a complex air dispersion model to map sources to concentrations at different locations. With only limited data, we are living in a second-best world where shortcuts are inevitable. Below we describe the simplified dispersion model used for our analysis.

Geographically localised ancillary benefits studies are able to incorporate more sophisticated dispersion models – e.g., of the Gaussian plume variety (see Colls 1997, ch.3, for a presentation of the Gaussian model with worked examples). This approach, which is rather data-intensive, is adopted in Cifuentes *et al.* (1999) for Santiago, Chile. From an economic perspective, the limitation of such studies is that climate policy is usually decided at national level by decision-makers interested in economy-wide costs and benefits which city-level or other local-level analyses cannot capture.

3.1. DISPERSION MODELLING

In the case where one is working with a single national CGE model, the question is what proportion of emissions from any given sector are having a significant impact on air quality, say, in a major metropolis. This is the problem we confronted in the case of Chile (Dessus and O'Connor 2001). While Santiago is not the only important city in Chile, it is by far the largest and suffers some of the severest air pollution. Thus, a decision was made to focus on Santiago and the national emissions numbers were scaled down, sector-by-sector, according to the share of a sector's output that is produced in the Santiago metropolitan area. Santiago emissions (say of particulates) were then linked to average yearly concentration via a linear dispersion equation, with the intercept term representing the background particulate level (i.e., the level that would occur “naturally”). Concentration data are available from readings taken by monitoring equipment. While emissions are generated by the model, a cross check against an independent emissions inventory showed broad consistency. The one missing link is exposure. To account for this, rather than a simple average annual concentration, we calculated a weighted average, where the weights for each monitoring station reading are the shares of the Santiago population living in the vicinity of that station (and the weights sum to 1).

$$\overline{CONC} = \frac{1}{n} \sum w_i \overline{CONC}_i, \text{ where } \sum w_i = 1$$

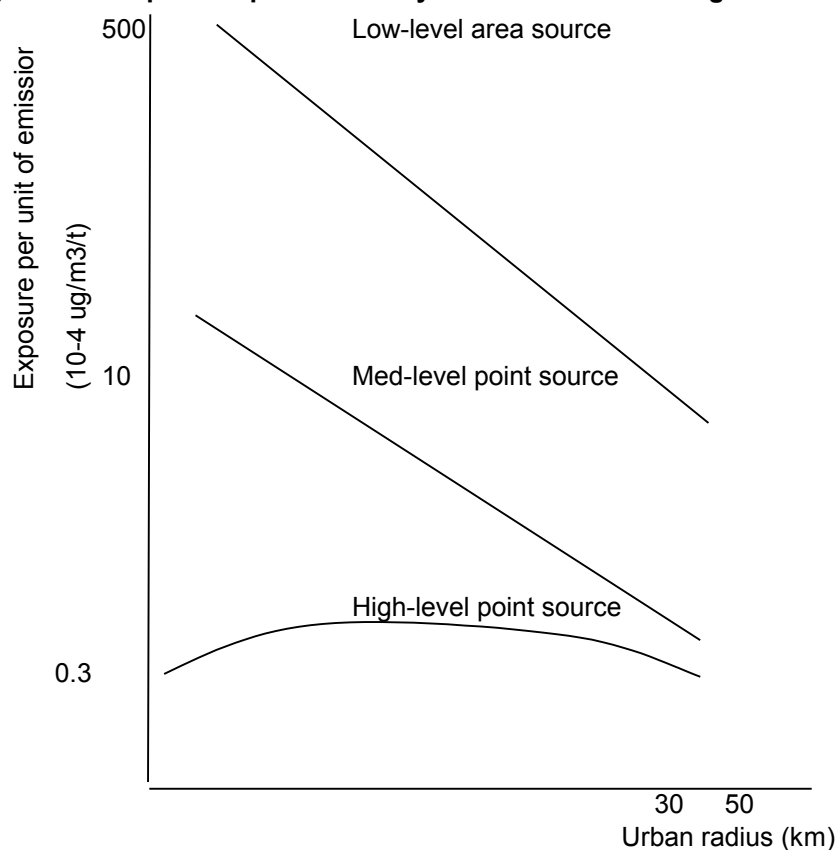
In the case of India, a large country with many polluted cities, a somewhat more sophisticated approach was taken. First, four regional economic models were constructed, giving rise to regional emissions. Then, these were linked to weighted average concentrations in each region's major city (or cities), using an equation like that above. Finally, rather than a single dispersion coefficient linking emissions to concentration, a model was used in which sources are differentiated by stack height (Garbaccio *et al.* 2000 employ a similar approach for China, but with a single national economic model). Economic sectors are grouped according to whether their emissions normally occur at or near ground level (e.g., small-scale industry, motor vehicles, household cooking), come from stacks of medium height (medium- to large-scale industry), or from high stacks (power plants) (classification based on Lvovsky and Hughes 1998). In short, the dispersion function (for particulates) is of the form:

$$CONC_{TSP} = a + b_1 (EMIS_{High}) + b_2 (EMIS_{Medium}) + b_3 (EMIS_{Low}),$$

where $CONC_{TSP}$ refers to the weighted average city-wide concentration and $EMIS_{High, Medium, Low}$ the region-wide emissions from each of three groups of sectors differentiated by typical stack height. The constant a is an approximation of the effect of background emissions on ambient concentration. The b_i s are the dispersion coefficients for emissions from each stack height, calculated using a simple dispersion model in which different atmospheric conditions are assumed to occur with given frequencies⁶ and the key piece of additional data required is a metropolitan area's radius (see WHO 1989 for the original model and Lvovsky *et al.* 1999 for an application in six cities). The use of even this somewhat more sophisticated dispersion model still involves a gross simplifying assumption, viz., that the specific geographic distribution of emission sources within the area does not significantly affect area-average pollutant concentration.

The WHO-type model yields the following results (see Figure 3.1): (i) for low and medium height sources, the concentration/exposure per unit of emissions is strictly inversely related to the city's radius – in other words, the wider the area over which emissions are dispersed, the smaller their effect on average ambient concentration; (ii) the emissions-exposure relationship for high-stack emissions follows an inverted-U shape in the city's radius, as high stacks contribute more widely to area emissions than low- or medium-stack emissions, so the contribution to area-average exposure rises at first with city size; and (iii) high-stack sources yield a concentration/exposure per unit of emissions very far below low-stack emissions for virtually any size of city and significantly below medium-stack emissions until city size approaches a radius of 30 km (in other words, a very large city). This suggests that the magnitude of any ancillary health benefits from changes in emission levels depends importantly on where (in which sectors) those changes occur.

Figure 3.1: Dispersion parameters by source emissions height



Source: WHO 1989

Wedding CGE models to adequate air dispersion models remains a research challenge, not least because even a regional CGE does not usually incorporate a detailed locational grid of emissions within the region of the sort needed for more sophisticated air modelling. To illustrate the problem, suppose that, while coal-burning power plants account for 50 per cent of regional particulate emissions and motor vehicles 20 per cent, the latter contribute 60 per cent to ambient concentrations in the main regional metropolis, while the former contribute only 30 per cent. Ideally, this locational effect on the emissions-concentration relationship should be reflected in the basic dispersion model, but without the benefit of a source-receptor matrix, one might mistakenly conclude that a 10 per cent reduction in power plant emissions would reduce concentrations and exposure in the big city by 5 per cent. Fortunately, although stack height represents a separate influence from geographic location in dispersion models, the former can to some extent proxy for the latter. This is because high-stack sources like power plants tend in general to be more remotely located from dense population concentrations than low-emission sources like automobiles, small industrial workshops, cooking stoves, etc. So, the lower coefficient on high-stack sources can be thought of as capturing in part the effect of remote location.

The linearity assumption in the dispersion model is perhaps reasonable for some pollutants, though not for all. In the case of particulates, it may be satisfactory as a representation of the primary dispersion of particulate matter, but it does not capture secondary particulate formation, which depends on the presence in the atmosphere of primary gases like SO₂ and NO₂, and ammonia (Colls 1997). Also, ozone (O₃) is the product of complex atmospheric chemical reactions between NO_x and VOCs in the troposphere.

3.2. DOSE-RESPONSE FUNCTIONS

Numerous dose-response studies exist linking air pollution to various endpoints – human health effects, agricultural productivity, forest growth, materials damage, etc. It is beyond the scope of this paper to review them (on human health effects, the reader is referred to Appendix D of EPA 1998). Reflecting the strength of the epidemiological evidence, the perceived relative health risks in developing countries, and the availability of monitoring data, our attention focuses on a relatively few air pollutants, with particulates by far the most important. In particular, the epidemiological literature finds a strong link between respirable (PM₁₀) and fine (PM_{2.5}) particle⁷ concentrations and respiratory-related mortality and morbidity.

The results from multi-city U.S. studies of acute exposure to PM₁₀ by Dockery, Pope and colleagues are quite consistent, finding an estimated 0.7-1.5 per cent increase in natural mortality associated with a 10 µg/m³ increase in PM₁₀ concentration from mean levels in the range 38-61 µg/m³ – i.e., several times lower than mean concentrations in many developing-country cities. A meta-analysis of multiple particulate-mortality studies done by Schwartz (1994) finds a consensus range for mortality increase estimates of between 0.7 and 1.0 per cent per 10 µg/m³ increase in PM₁₀ concentration. Comparing their estimates to those of other studies, Dockery *et al.* (1992) observe that the dose-response relationship between particulates and mortality is remarkably similar across a large range of concentrations, in a variety of communities, and with varying mixtures of pollutants and climatology. There is no evidence of a “no effects”, or threshold, concentration – at least not within the range observed in U.S. cities.

The robustness of the estimates is borne out by non-U.S. studies, including a handful in developing countries. For instance, Ostro *et al.* (1996) find a significant relationship, for Santiago, Chile, between ambient PM₁₀ concentration and mortality after controlling for confounding influences like temperature. In particular, the results from their basic OLS model suggest that a 10 µg/m³ change in concentration around the mean (115 µg/m³) is associated with a 0.6 per cent change in mortality⁸. Similarly, Chesnut *et al.* (1997) find relative mortality risk changes in Bangkok, Thailand, very similar to those from U.S. studies. The consistency of results between the U.S. and certain developing countries suggests the possibility of transferring dose-response function coefficients in cases where local studies are not available, assuming the target location is not drastically different from the United States in terms of variables like time spent outdoors, baseline health status, and medical care and access.

Besides mortality, there are a variety of morbidity endpoints that may be affected by air pollution, though few relationships are borne out as consistently by the epidemiological literature as the PM_{10} – mortality link. There are two main ways in which the effects of pollution on morbidity are measured: as incidence of physical symptoms and illness and as behavioural responses to the symptoms/illness. The former are normally the object of interest in clinical studies, while epidemiological studies may report on symptom/disease incidence and/or effects on human activity. The most common measures for the latter are “restricted activity days” (RADs), “work loss days” (WLDs), acute respiratory symptom days, respiratory hospital admissions (RHAs), and emergency room visits. RADs are a more comprehensive measure than WLDs, including days spent in bed, days missed from work, and other days when normal activities are restricted due to illness (Cifuentes and Lave 1993). They are also a more subjective measure and thus susceptible to greater measurement error.

The dose-response functions for morbidity endpoints appear to be more variable between developed and developing country sites than those for particulate-related mortality. Thus, the Chesnut *et al.* study finds that the central estimates of relative risk are roughly comparable for RHA between Bangkok and the United States, cases per million population differ considerably, with Thailand’s lower number reflecting the weaker propensity to seek hospital care for respiratory symptoms. On the other hand, the number of acute respiratory symptom days is far higher in Bangkok than in the United States. Thus, for these morbidity endpoints, dose-response coefficient transfer across countries could be more problematical than for mortality.

Table 3.1. provides central estimates, based on the available epidemiological evidence, of the slope coefficients of the dose-response functions linking major air pollutants to various health endpoints. These are the dose-response relationships incorporated in the models for Chile and India, though in the Indian case concentration data were only available for the first three pollutants. In Dessus and O’Connor (1999), lead was included in the analysis for Chile, but in their more recent paper (2001) it is not.

Table 3.1: Dose-Response Function Slopes (Central Estimates)

	PM-10 (10 µg/m ³)	SO ₂ (10 µg/m ³)	NO ₂ (pphm)	CO (ppm)	Ozone (O ₃) (pphm)	Lead (Pb) (µg/m ³)
Premature mortality/100,000 pop.	6.72				0.006	
Premature mortality/million males age 40-59						350
RHA/100,000	12				7.7	
ERV/100,000	235					
RAD/person	0.575					
MRAD/person					0.34	
Clinic visits for LRI/child age < 15	0.0028					
Respiratory symptoms/person	1.83				0.55	
Respiratory symptoms/adult			0.10			
Respiratory symptoms/1,000 children	16.9	0.18				
Asthma symptoms/asthmatic	0.33				0.68	
Chronic bronchitis/100,000 age > 25	44					
Chest discomfort/adult		0.10				
Eye irritation/adult					0.266	
Headache/person				0.013		
IQ decrement						0.975
Hypertension/million males age > 20						72,600
Non-fatal heart attack/million males age 40-59						340

Note: For blank cells, there is no known significant relationship between the pollutant and health endpoint.

Sources: Ostro (1994); Ostro *et al.* (1998); Schwartz and Dockery, 1992; World Bank (1994).

4. VALUING ENVIRONMENTAL IMPACTS

There are three broad approaches to valuation of environmental benefits in general and ancillary benefits of climate policies in particular (see Freeman, 1993, for the classic text on valuation methods). The first approach tallies productivity losses or costs to the economy from illness, premature death, or damage to crops, materials and ecosystems. From a theoretical standpoint it is the least satisfactory, not being firmly grounded in welfare economics. It can, however, provide a lower bound estimate of “true” benefits. The other two approaches – the first based on revealed preferences and the second on stated preferences – are more firmly grounded on individuals’ utility functions, but they are not without their own measurement problems.

In most health effects valuation exercises, mortality benefits tend to dominate morbidity ones, largely because of the high value individuals attach to reducing the risk of premature death. As explained above, in our analysis we rely wherever possible upon stated or revealed preference methods of evaluating a statistical life.

The problem the developing country researcher often faces is the paucity (often the total absence) of studies estimating *VSL* for his or her own country, hence, the need to rely on “benefits transfer” from sites more frequently studied, most commonly the United States. That is only the beginning of the problem, however, as the *VSLs* estimated in different U.S. studies are “all over the map”. Thus, in a review of 26 studies valuing mortality benefits, Viscusi (1992) finds that the lowest and highest estimates of *VSL* differ by a factor greater than 20 – ranging from \$600,000 to \$13.5 million (at 1990 prices). It is possible to fit a distribution to these estimates (as done in Appendix I of EPA 1998, with a Weibull distribution offering the best fit). Still, while the mean value is \$4.8 million per premature death avoided, the standard deviation is \$3.24 million, suggesting a wide margin for uncertainty in “benefits transfer”.

Most of the U.S. studies from which *VSL* estimates are derived calculate compensating wage differentials for higher on-the-job exposure to the risk of fatality. From such hedonic wage estimation one can derive an estimate of *VSL*. For instance, if it is found that, on average, a worker receives a wage differential of \$350 per year for assuming an added risk of accidental death on the job of 1/10,000, then this implies a *VSL* of \$3.5 million. When one transfers this estimate out of the context in which it was derived – e.g., to one of mortality risk from pollution – there are at least four possible sources of bias, two having to do with different risk characteristics and two with different risk preferences of affected groups of individuals. First, assuming complete information, job-related risk is voluntarily assumed, while risk from pollution exposure is largely involuntary. Second, the time dimension of the risks can differ. For example, certain risks from pollution exposure are delayed until later in life, and people may value differently risks avoided now to those avoided later (see Krupnick 2001). With respect to the current analysis, this is not a serious problem, since we are concerned principally with the short-term effects of acute exposure to particulates and other air pollutants, not so much with long-term effects of chronic exposure.

Third, populations may differ in their attitudes towards risk. There may be some selection bias in hedonic wage studies if risk-averse individuals self-select into safer jobs. In that event, the results of such studies would provide a downwardly biased estimate of *WTP* for risk reduction of the entire working age population.

This raises a fourth issue, which is that the age profile of the sample used in a hedonic wage study may not be very representative of the age profile of the high-risk population from air pollution. In the United States, the high-risk groups are primarily the elderly, the infirm, and the very young. In the case of India, Cropper *et al.* (1997) find for Delhi that, while the overall particulate-related mortality risk is only about one-third of that found in U.S. studies⁹, the age group at greatest risk is also quite different: the 15-44 age group *versus* those over 65 years in the United States. Clearly, this can make a difference to

calculation of life-years lost, though how big a one depends on the difference in life expectancies between the United States and a particular developing country¹⁰. As for contingent valuation of *VSL*, however, Krupnick (2001) and his colleagues find for a sample of Canadians that, below the age of 70, age does not seem to matter to willingness to pay for reduced mortality risk¹¹. Those in the 70-75 age group were willing to pay one-third less than the average for a given reduction in mortality risk. Since in developing countries the number of people who survive to that advanced age is still rather small (in Delhi, e.g., 70 per cent of deaths occur before the age of 65), the use of *VSL* estimates from random population samples would not appear to introduce serious bias.

Beyond these four issues in *VSL* transfer across contexts and populations, transfer of *VSL* estimates from a developed country source site to a developing country target site poses another issue – viz., how to adjust for differences in *per capita* income, hence in ability to pay. One needs to scale the *VSL* for these differences, but a simple ratio of *per capita* incomes would only be the appropriate scaling factor if the income elasticity of *VSL* (or more broadly *WTP*) is equal to unity. The formula for calculating *WTP* in a target developing country, given a *WTP* (or *VSL*) estimate for a wealthy source country like the United States is:

$$WTP_T = WTP_S (1 - \xi_{wtp,y}(Y_S - Y_T)/Y_S),$$

where T and S subscripts refer to the target country and the source country, respectively, Y stands for *per capita* income, and $\xi_{wtp,y}$ is the elasticity of *WTP* with respect to *per capita* income. What empirical evidence is there on the size of this elasticity? As with the *VSL* itself, estimates vary fairly widely. Until recently, the few studies available suggested that the elasticity is positive but less than one: in other words, survival is a necessity. In their benefits transfer study of air pollution in Central and Eastern Europe, Krupnick *et al.* (1996) assume an elasticity of 0.35 (based on contingent valuation studies reported in Mitchell and Carson 1986). Studies estimating *WTP* for pollution-related morbidity benefits also find an income elasticity less than unity, with Loehman and De (1982) estimating a range from 0.26 to 0.60 for reduced respiratory symptoms from cleaner air in Tampa, Florida, Alberini *et al.* (1997) an elasticity of 0.45 for similar benefits in Taiwan, and Liu *et al.* (2000) an elasticity of a mother's *WTP* to prevent a cold of 0.3 (for her child) to 0.4 (for herself). On the other hand, Hammitt *et al.* (2000) find, from a longitudinal (16-year) compensating wage differential study for Taiwan, that “survival is a luxury good”, estimating an income elasticity of *VSL* of between 2 and 3. A meta-analysis of *VSL* studies by Bowland and Beghin (1998) yields a similar elasticity estimate.

Why is the choice of $\xi_{wtp,y}$ important? A simple numerical example makes this clear. Suppose one is planning to transfer the *VSL* value mentioned above from a meta-analysis of U.S. studies (EPA 1998) to China. In 1998, China's PPP *per capita* income was 1/10th that of the United States. So, if we were to assume $\xi_{wtp,y} = 1$, then 1/10 becomes the adjustment factor. Since the U.S. *VSL* is \$4.8 million, China's must be \$480,000. What happens if instead we use $\xi_{wtp,y} = 0.5$ or $\xi_{wtp,y} = 2.0$. In the former case, we have by the above formula (with monetary values expressed in '000 US\$):

$$WTP_T = 4,800 [1 - 0.5(29.6 - 3.1)/29.6] = 4,800 - 2,149 = 2,651$$

In the latter case, we have:

$$WTP_T = 4,800[1 - 2.0(29.6 - 3.1)/29.6] = 4,800 - 8,595 = - 3,795,$$

which is a nonsensical result. This is because, mathematically,

$$WTP_T \Rightarrow 0 \text{ as } \xi_{wtp,y} \Rightarrow Y_S / ((Y_S - Y_T))$$

So, in this case, any value of $\xi_{wtp,y} > 1.11$ would result in negative WTP for China.

Table 4.1 (borrowed from Dessus and O'Connor 2001) provides an illustration of how different values of $\xi_{wtp,y}$ would translate into different benefits transfer values for various health endpoints between the United States and Chile.

Table 4.1: Estimated Monetary Values of Unit Changes in Various Health Endpoints

Health Endpoint	United States	Chile			Units	Estimation Method
		elasticity = .5	elasticity = .75	elasticity = 1		
	1992 US\$					
Respiratory hospital admission (RHA)	7,058	4,376	3,035	1,694	\$/event	Cost of illness(COI)
Emergency room visit (ERV)	199	123	86	48	\$/event	COI
Restricted activity day (RAD)	57.5	35.7	24.7	13.8	\$/day	Wages foregone
Minor restricted activity day (MRAD)	24.3	15.1	10.4	5.8	\$/day	Contingent valuation (CV)
Chronic bronchitis in adults	237,604	147,314	102,170	57,025	\$/case	CV
Asthma attack	33.4	20.7	14.4	8.0	\$/attack day	CV
Respiratory symptom day	6.7	4.2	2.9	1.6	\$/day	CV
Child respiratory symptom day	5.4	3.3	2.3	1.3	\$/day	CV
Adult chest discomfort case	6.7	4.2	2.9	1.6	\$/event	CV
Eye irritation	6.7	4.2	2.9	1.6	\$/event day	CV
Headache episode (avg. of mild and severe)	27.2	16.9	11.7	6.5	\$/event day	CV

Notes:

1992 PPP exchange rate: 186 Chilean pesos/USD

Ratio: PPP per capita income (Chile/US), 1992: 0.24

Sources: Krupnick *et al.* 1996; EPA 1998, Appendix D; Beghin *et al.* 1999.

Another way in which the elasticity matters is when, as here, the analysis is dynamic, involving simulations over future time. Clearly, as *per capita* income rises in the simulation, how fast VSL will rise depends critically on the assumed value of $\xi_{wtp,y}$.

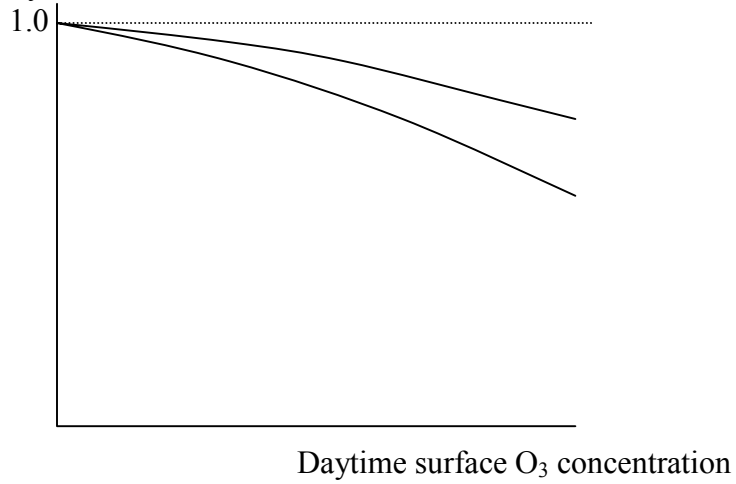
While the discussion thus far has focused on the valuation of health impacts, other impacts may be of interest. In our ongoing study of China¹², for example, the focus is on agricultural yield impacts of altered tropospheric ozone and particulate haze levels as a result of climate policy. In this case, the valuation of impacts is rather straightforward, in that agricultural commodities are priced in markets, so any yield effects translate directly into changes in quantities exchanged and possibly also prices. Even this analysis is complicated, however, by so-called feedback effects. Thus, if local air pollution reduces crop yields, then a policy that lowers that pollution will improve those yields and, if relative prices are affected, this will in turn alter resource allocation across sectors. In effect, the policy lowers an external cost to farmers of growing their crops and, assuming that this cost reduction is not fully offset by the combination of reduced output prices and increased input costs as a result of the carbon tax, farming becomes more profitable. This in turn should result in a changed resource allocation, with more resources flowing into the farming sector. A change in economic structure, however, will also lead to a change in emissions levels, which will in turn affect agricultural yields. The likelihood is that, if resources shift into agriculture from elsewhere in the economy, this will lower air pollution (since agriculture is among the least-air-polluting sectors), having a further positive effect on yields. Whether (and how quickly) this process converges depends on three things: the shape of the crop dose-response curves, how marginal input costs change with the tax, and the elasticities of the crop demand and supply curves.

A graphical exposition is given in Figure 4.1. If there are diminishing returns to additional improvements in local air quality (e.g., to a reduction in tropospheric ozone levels), this implies that the crop yield curve is non-linear, as shown in the first panel (based on Figure 3 of Aunan *et al.* 2000; see also Chameides *et al.* 1999). Thus, the marginal improvements in yield decline for each additional unit reduction in pollution. Then, the supply curve for a particular commodity will shift out proportionately less for each incremental reduction in pollution (Panel B). If marginal input costs are also increasing in pollution abatement (or, put differently, in the carbon tax – see Figure 1.A above), then each incremental unit of abatement would cause a proportionately bigger leftward shift of the supply curve (Panel C). At some point, the two forces – of yield improvements at a diminishing rate, and of input cost increases at an increasing rate – would balance each other, with the result that at that abatement rate there would be no further shift in the crop supply curve and the economy would settle into a new equilibrium. Exactly where the new equilibrium would be – with what crop price/quantity combination – will in turn depend on the elasticities of supply and demand.

Figure 4.1: Crop Yield Effects of Pollution Reduction in an Economy-wide Model

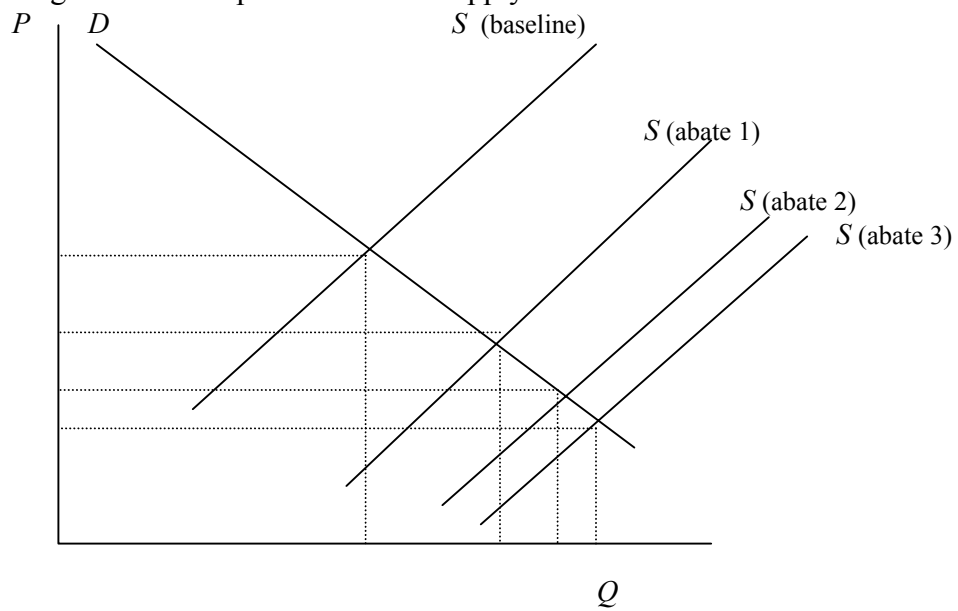
Panel A

Relative yields



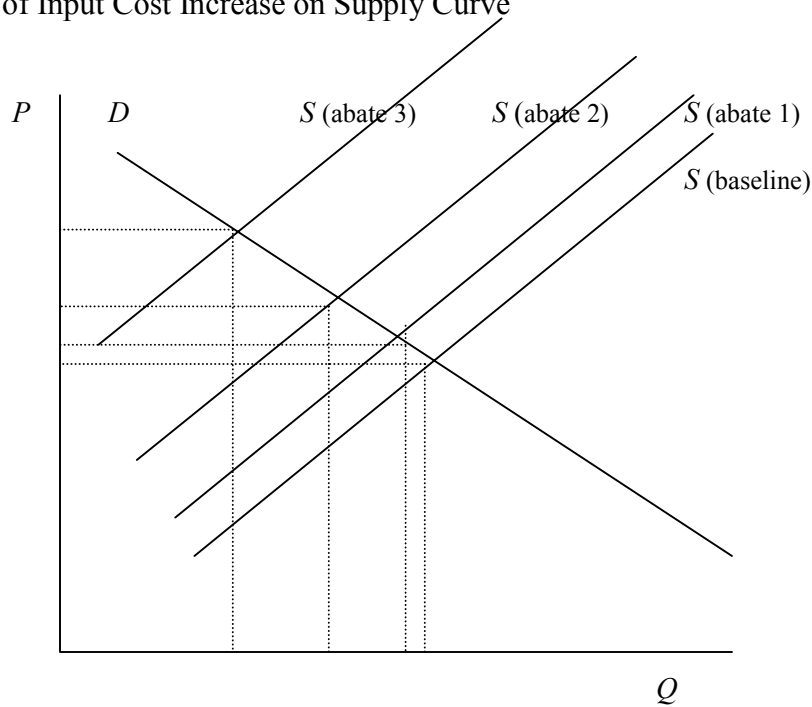
Panel B

Effect of Marginal Yield Improvements on Supply Curve



Panel C:

Effect of Input Cost Increase on Supply Curve



Note: (abate 1) = (abate 2) = (abate 3) in volume terms.

In the cases of both health impacts and agricultural crop yields, the noted ancillary benefits are only part of the story and, depending on the time horizon of the analysis, the other part of the story may be more or less important. That other part consists in the effects on health and on agriculture of climate change itself, hence of averting climate change. Looking out a decade to the first Kyoto commitment period, these primary effects of climate change are not likely to be easily detectable and quantifiable. What is even more certain is that, in that time horizon, the effects on global temperature and climate of any interim measures to curtail GHG emissions would be vanishingly small. So, for the purpose of short-term analysis, they can safely be ignored.

5. BASELINE AND POLICY SIMULATIONS: ESTIMATING SCOPE FOR “NO REGRETS”

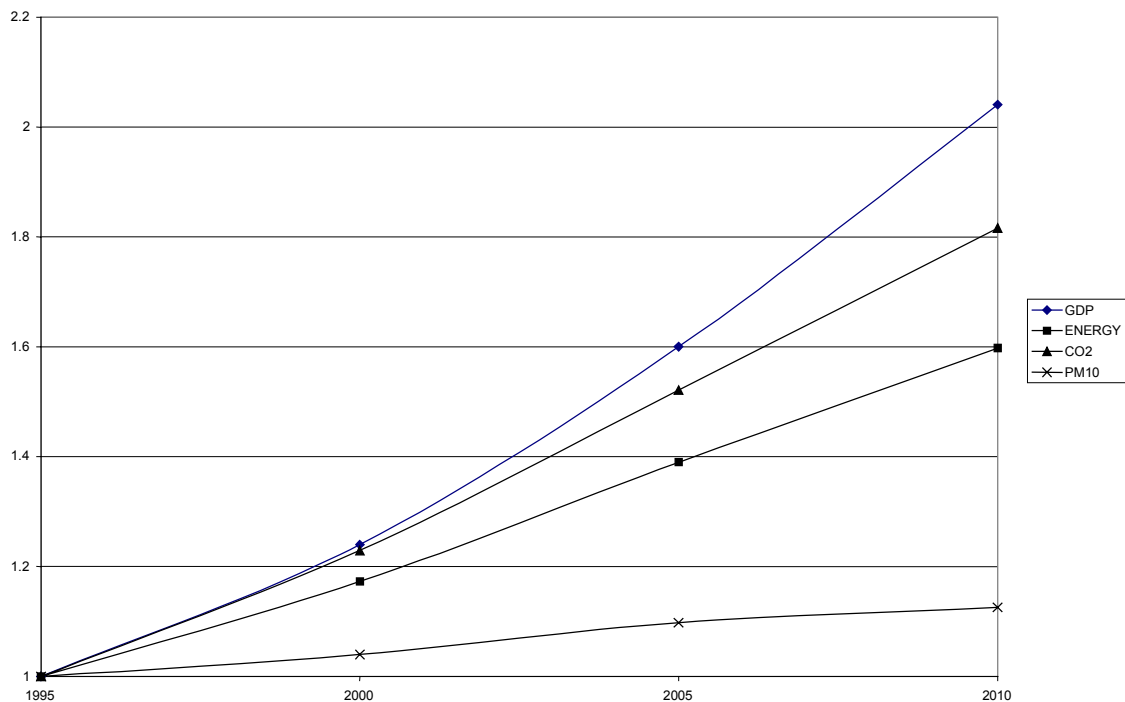
We are now ready to begin the model simulations, first establishing a plausible growth baseline and then conducting policy experiments in terms of deviations from emissions baseline.

5.1. THE BASELINE SIMULATION

The construction of a plausible baseline requires, for a start, a reasonable degree of certainty about mid-term economic prospects. This is fairly easy for countries characterised by long periods of macroeconomic stability, but much more difficult for those vulnerable to shocks. In the long-run, one could simply focus on trend growth, abstracting from cyclical fluctuations, but in a decadal timeframe shocks can matter – witness Latin America’s “lost decade” following the Mexican debt crisis of the early

1980s. For countries like China and India, where macroeconomic instability is less of a problem, projecting GDP growth for a decade does not pose insurmountable problems. Other important assumptions include the evolution of energy consumption in relation to GDP. As explained above, this is dealt with – albeit mechanically – by the assumption of a certain rate of autonomous energy efficiency improvement (AEEI), usually one per cent per annum. The most problematic is what to assume about environmental policy evolution over the scenario period. Will government become more determined and effective in its efforts to control local air pollution, and if so by how much? Inevitably, the assumption one must make on this score is somewhat arbitrary, though past performance can provide some guidance. Thus, if particulate emissions have continued to rise despite the imposition of stringent emission standards, it may be a fair guess that they will continue to do so for some time. Nevertheless, a useful rule of thumb would be to give the government the benefit of the doubt when establishing a local emissions baseline. In this way, one cannot be accused of “loading the dice” in favour of large ancillary benefits. An example of such a “conservative” baseline is given in Figure 5.1 for Chile, conservative notably in that particulate (PM₁₀) emissions are assumed to rise very slowly to 2010. One noteworthy feature of this baseline is that CO₂ emissions rise faster than energy consumption. This is not a general phenomenon across countries but reflects the initial heavy reliance of Chile on hydroelectric power, with the expectation that spare hydro capacity will be able to meet only a small share of the country’s incremental electricity demand. Thus, a larger share will be generated by fossil fuels.

Figure 5.1: Chile: Baseline Trends in GDP, Energy and Emissions

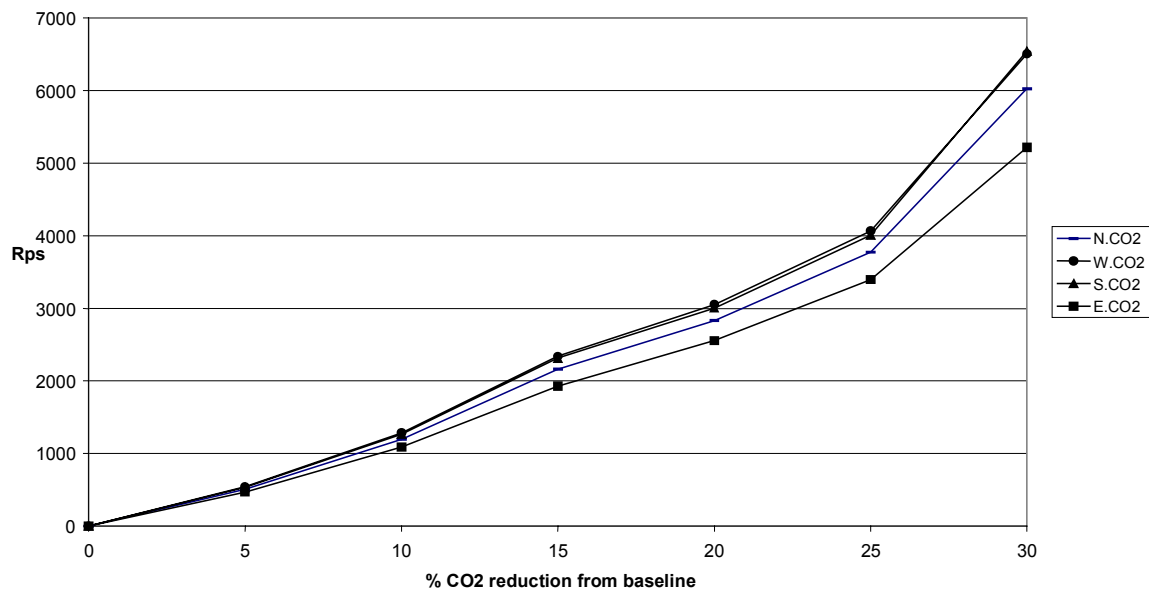


Source: Dessus and O’Connor (2001).

5.2. POLICY EXPERIMENTS AND SENSITIVITY ANALYSIS

The policy experiment consists in constraining CO₂ emissions to be some given percentage below baseline emissions by 2010. The means of constraining emissions is, as explained above, the imposition of a national tax based on the carbon content of fuels. It was also noted earlier that national authorities would normally be the ones to decide on climate policy like a carbon tax, so the choice of a single national tax instrument corresponds most closely to political realities. It is not an entirely innocuous choice, however, as the India study shows. There, there are four regional economic models, with abatement costs and ancillary benefits both calculated at the regional level. These can, and do, vary across regions. Microeconomics tells us that, for cost minimisation, abatement costs of different economic actors (whether firms or in this case regions) should be equated at the margin. The bigger the cost differences across regions, moreover, the greater the cost savings from using a common tax rate *versus* having region-specific emission targets and tax rates. Figure 5.2 shows the regional marginal abatement cost curves, as given by the carbon tax required to achieve a given percentage CO₂ reduction from baseline, region by region. As can be seen, the E region has the lowest abatement costs and W and S the highest. So, with a single national tax, E will abate proportionately more CO₂ and W and S proportionately less. This solution would minimise national abatement costs.

Figure 5.2: CO2 Tax per Tonne of CO2, Region-Specific Targets



Cost-effectiveness, Optimality and Equity

This is not the end of the story, however, since ancillary benefits also vary considerably across regions, reflecting differences in the weights of different types of source and differences in population exposure. Thus, an optimal allocation of abatement effort would be one in which each region was abating to the point where its regional marginal costs were equal to regional marginal (\equiv average) ancillary benefits. Thus, if the national government were to set a common national carbon tax, and if it were concerned with

interregional equity, it would need to use some of the revenue from the tax to compensate the “loser” – in this case, the South which, as Table 5.1. shows, does not appear to enjoy any net benefits from such a tax even at the low abatement rate of 5 per cent. To complicate matters further, the Table also points to another problem, viz., that even if a region is one of the larger net beneficiaries of the policy, when ancillary benefits are valued, it may be a relatively large loser in terms of conventionally measured disposable income (as for example with E). Since disposable income shows up in national income statistics and ancillary benefits do not, policy makers might well have a hard sell to convince E that it ought to agree to compensate S for the latter’s welfare losses.

Table 5.1: Welfare Costs and Net Benefits, by Region of India- 2010

		<i>Reduction in CO2 emissions % (Final year Simulation wrt Final year BAU)</i>					
		5	10	15	20	25	30
As % of Real GDP							
Welfare Costs							
Nor.EV		-0.08	-0.19	-0.37	-0.76	-1.25	-1.64
Wes.EV		-0.11	-0.27	-0.50	-0.95	-1.53	-2.04
Sou.EV		-0.10	-0.25	-0.47	-0.89	-1.43	-1.89
Eno.EV		-0.13	-0.29	-0.52	-0.96	-1.51	-1.96
Total		-0.10	-0.25	-0.46	-0.89	-1.43	-1.89
Net Benefits							
Nor.NetBenefits		0.27	0.49	0.64	0.57	0.39	0.32
Wes.NetBenefits		0.04	0.03	-0.06	-0.37	-0.79	-1.16
Sou.NetBenefits		-0.01	-0.06	-0.19	-0.52	-0.97	-1.33
Eno.NetBenefits		0.14	0.23	0.25	0.06	-0.25	-0.44
Total		0.11	0.16	0.14	-0.09	-0.44	-0.70
% Change in:							
Disposable Income (After Taxes)							
Nor		-0.26	-0.61	-1.10	-1.58	-2.21	-3.10
Wes		-0.38	-0.88	-1.54	-2.17	-2.98	-4.23
Sou		-0.20	-0.49	-0.90	-1.36	-1.96	-2.75
Eno		-0.62	-1.36	-2.29	-3.13	-4.17	-5.67
Real GDP BAU Shares							
Nor	25	-0.11	-0.26	-0.45	-0.75	-1.14	-1.50
Wes	34	-0.12	-0.28	-0.49	-0.81	-1.21	-1.61
Sou	24	-0.11	-0.26	-0.45	-0.76	-1.15	-1.52
Eno	17	-0.15	-0.34	-0.57	-0.91	-1.33	-1.71
India	100	-0.12	-0.28	-0.48	-0.80	-1.20	-1.58

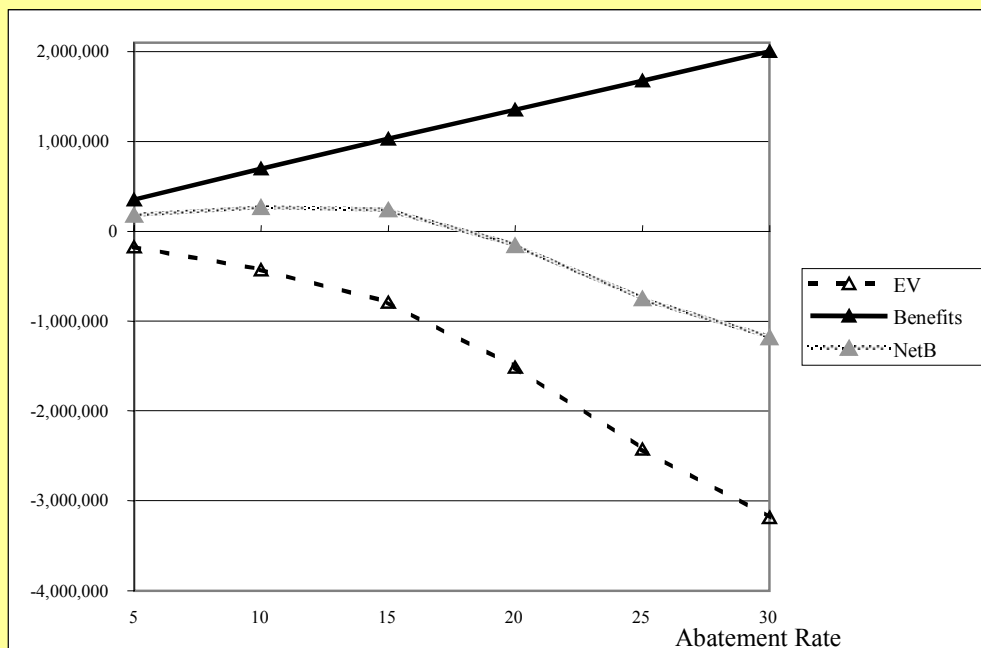
Another aspect of equity that governments will almost certainly wish to consider is the effect of the policy on inter-household income distribution. This requires, however, that the analysis incorporate more than a single representative household. For that, household expenditure survey data is needed both to group households into expenditure classes, from the lowest to the highest, and to differentiate patterns of expenditure by expenditure class. Short of such detailed analysis, a few tentative observations are possible, based on what the model results show of the changes in labour income and capital income respectively as a result of the carbon tax. Household income is derived largely from factor ownership, with labour being the principal if not sole source of income for the poor and capital figuring more prominently in the incomes of the rich. The policy simulation shows that, while many sectors contract as a result of the carbon tax, a few expand, and these tend to be among the most labour-intensive ones (agriculture, food processing,

textiles and clothing). *Prima facie*, this would suggest that the relative demand for labour is rising and, *ceteris paribus*, its relative return. This is indeed the case. *Relative* needs to be stressed, as incomes earned by both capital and labour decline with the carbon tax. Capital income, however, declines at a slightly faster rate than labour income, with the result that the wage/rental ratio rises by about 1.4 per cent for a 15 per cent CO₂ reduction. So, from this perspective, a carbon tax is mildly progressive. On the expenditure side, of course, the story could be different if, for example, poor households spend a higher proportion of their income than the rich on coal and other products with a high coal content. There is also the question of the incidence of the ancillary benefits. Here, if – as seems plausible – the poor tend to be more heavily exposed to outdoor air pollution than those in the middle to upper income groups, then they should capture the bulk of the benefits from cleaner air. The net distributional effects are ambiguous in the absence of further analysis.

No Regrets Abatement and Its Decomposition

Figure 5.3 illustrates the “no regrets” rate of abatement for India in 2010 (using central estimates of substitution elasticities and of *WTP* for health improvements). It corresponds to a 15+ per cent reduction in emissions from the baseline (which implies that, instead of growing by about 80 per cent between 1995 and 2010, CO₂ emissions would grow by only about 50 per cent). Does this mean, then, that policy makers can confidently impose a carbon tax calculated to reduce baseline 2010 emissions by 15 per cent? It would if we were living in a world of perfect information and zero uncertainty. ... In practice, not. Hence the need for sensitivity analysis, to give policy makers a broader range of options, depending on their degree of risk aversion.

Figure 5.3: No Regrets CO₂ Abatement, India (10⁵ Rps)



Note: EV stands for Equivalent Variation, and NetB for Net Benefits

We can decompose the CO₂ emission reductions, either for the national economy or for each region (in the case of India), into several components, including changes in: (i) the sectoral composition of output, (ii) the carbon-intensity of energy, (iii) the energy intensity of the economy, and (iv) the scale of production. Consider the following identity, which simply states that total emission is equal to the sum of sectoral emissions:

$$E = \sum_i \left(\frac{X_i^{Output}}{X_{tot}^{Output}} \frac{E_i}{Ene_i} \frac{Ene_i}{X_i^{Output}} X_{tot}^{Output} \right)$$

where E is total emission volume, X_{tot}^{Output} total output (in real terms), E_i the sectoral emission volumes, Ene_i the sectoral fuel (energy) use, and X_i^{Output} the sectoral outputs. The first term on the right corresponds to (i), representing sector i 's share in total output; the second term captures (ii), the carbon emissions intensity of energy consumption; the third term (iii), the energy intensity of sectoral production; the fourth term (iv), the scale of the economy. The total variation in emission levels can then be measured as the sum of the mentioned four components by differentiating the above identity:

$$\partial E = \sum_i \left[\partial \left(\frac{X_i^{Output}}{X_{tot}^{Output}} \right) \frac{E_i}{X_i^{Output}} + \partial \left(\frac{E_i}{Ene_i} \right) Ene_i + \partial \left(\frac{Ene_i}{X_i^{Output}} \right) \frac{E_i}{Ene_i} X_i^{Output} + \partial (X_{tot}^{Output}) \frac{E_i}{X_i^{Output}} \right]$$

Table 5.2 shows the decomposition for the four regions of India. What is clear is the dominant role played by a reduction in carbon-intensity of energy, through inter-fuel substitution and a change in the energy mix. Improved energy efficiency of the economy is usually the next most important source of emission reductions, with the exception of the E region where shifts in the sectoral composition of output weigh more heavily. This is presumably because of the heavy dependence of the latter on the coal mining sector and energy-intensive industries that go into relative decline. Thus, resources must shift out of this sector into others. The large scope for energy savings is consistent with studies that find India's heavy industries (e.g., iron and steel) to be well behind the world efficiency frontier¹³.

Table 5.2: Decomposition of CO2 Emission Changes, India by Region

5 <i>Abatement rate (%)</i>				
<i>Share of total CO2 reduction attributable to change in:</i>	Nor	Wes	Sou	Eno
Sectoral composition	10.2	6.3	7.0	21.6
Carbon-intensity of energy	65.9	71.2	68.5	55.7
Energy-intensity of output	17.4	16.2	18.7	16.2
Scale of production	6.4	6.3	5.9	6.5
10				
	Nor	Wes	Sou	Eno
Sectoral composition	11.3	7.1	7.7	23.2
Carbon-intensity of energy	65.2	70.2	68.2	54.7
Energy-intensity of output	16.7	15.9	17.7	15.2
Scale of production	6.8	6.8	6.3	6.9
15				
	Nor	Wes	Sou	Eno
Sectoral composition	11.6	7.4	7.9	23.5
Carbon-intensity of energy	65.1	69.6	68.2	54.9
Energy-intensity of output	16.1	15.6	17.1	14.4
Scale of production	7.3	7.4	6.8	7.2

Uncertainty and Sensitivity Analysis

Sensitivity analysis is performed to determine how dependent results are on the particular values of certain key parameters and exogenous variables used in the simulations. There are any number of sensitivities one could test, but theory-informed intuition is used to select the few thought to be most important. These are the substitution elasticities¹⁴ in the production function and the *WTP* values. (A proto-sensitivity-analysis was performed above for the elasticity of *WTP* with respect to income – see Table 4.1, so that parameter is not treated further.) In a longer-term exercise, one would also be interested in testing alternative time paths of emission reductions. It is worth noting that the introduction of ancillary benefits into the analysis of climate policy may significantly alter the standard result that “it pays to wait” because of the prospect of future technological improvements that will lower the costs of abatement. Now, these benefits of waiting need to be weighed against the costs in terms of additional lives lost and poor health in the here and now.

Clearly the fact that we employ sensitivity analysis at all suggests that we are uncertain about the “true” values of certain parameters. Thus, in conducting the analysis, it would make sense to choose alternative values that bound a range of plausible estimates. Ideally, this would involve specifying a statistical distribution such that one can assign a specific probability to observing parameter values within the chosen range (e.g., 95%, 99%). This is not always possible (requiring at a minimum information about the standard deviation around a sample mean). In that event, more arbitrary rules can be used – e.g., multiplying substitution elasticity values by 0.5 and 1.5. This is what was done in the case of India. In the case of *VSL*, the lower and upper bounds of the range of plausible

India-specific estimates¹⁵ was taken for the sensitivity analysis (the mid-point of that range being the central *VSL* value employed) (see Bussolo and O'Connor 2001).

Figure 5.4 presents the combined results of a series of sensitivity analyses in which, first, substitution elasticities in production were allowed to vary, holding *WTP* constant at central values, then *WTP* was varied, holding substitution elasticities constant at default values, then both substitution elasticities and *WTP* were allowed to vary. The “low elasticity, low *WTP*” results in the first panel are the least favourable to “no regrets” climate policy, while the “high elasticity, high *WTP*” results are most favourable. The dashed vertical lines through the horizontal axis show the “no regrets” abatement rates associated with each. Even in the least favourable case, the “no regrets” abatement rate is in the 10-15 per cent reduction range relative to baseline 2010 emissions. Ten per cent, then, would appear to be a “safe lower limit” on CO₂ abatement for a risk-averse Indian policy maker. It should be noted that, if we were to include other greenhouse gases in the analysis, the “safe lower limit” would in all likelihood be raised insofar as other gases offer low cost abatement options over some range¹⁶.

5.3. COMPARISON WITH OTHER RESULTS

We have presented the results generated by the CGE-based policy simulations, but are these reasonable? What is our point of reference? There are several markers one can use for cross-checking results, to see if they fall within a plausible range. (A *caveat* of such comparisons is that it might just be that all the previous studies produced erroneous results.) The most common metrics for the economic analysis of climate policy are: the marginal cost of reducing a tonne of carbon (tC) (hence, the carbon tax rate) and, in the case of ancillary benefits, the size of those benefits per tC reduction. The latter, in turn, can be measured either in physical units (e.g., premature deaths avoided or, put simply, lives saved per unit C reduction) or in monetary units. Our focus here is on the ancillary benefits comparison.

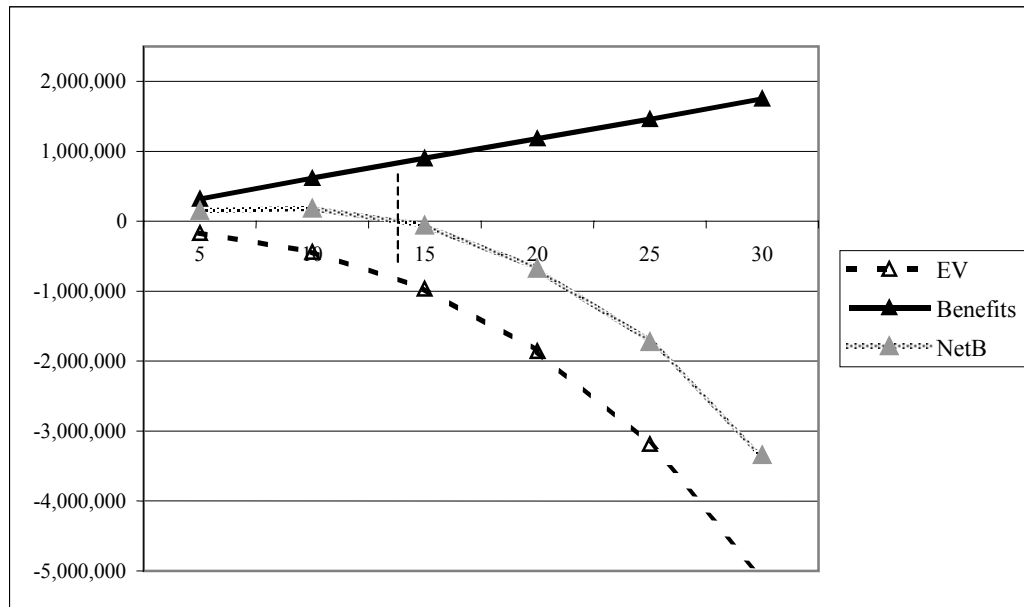
In making cross study (and especially cross-country) comparisons of ancillary benefits, it is important to bear in mind the several possible sources of variance in results, including differences in: underlying parameter values or variable estimates – e.g., of *VSL*; method of estimating benefits (e.g., damages avoided *versus* abatement costs avoided); scope of benefits included (e.g., some studies include both emission-related benefits and non-emission-related ones like reductions in traffic congestion, accidents, and noise resulting from reduced road transport); population exposure to pollution across study sites.

Table 5.3 summarises calculations based on several studies that have looked at mortality benefits from climate policy. Our results show lives saved per million tonnes of carbon abated equal to 334, compared with 298 for China estimated by Garbaccio *et al.* The numbers for Chile and the United States are considerably lower. These relative magnitudes are broadly consistent with the hypothesis state above that developing countries with few initial local pollution controls (hence, little delinking of CO₂ emissions from other pollutants) are likely to benefit more climate policy – in terms of lives saved per tC reduction – than developed countries where such delinking is more

advanced. Another explanation for the high number of lives saved in China and India is the high urban population densities, hence, the large exposed populations relative to Chile and the U.S.A.

Figure 5.4: Outer Bounds on "No Regrets"

Low Elasticities, Low WTP



High Elasticities, High WTP

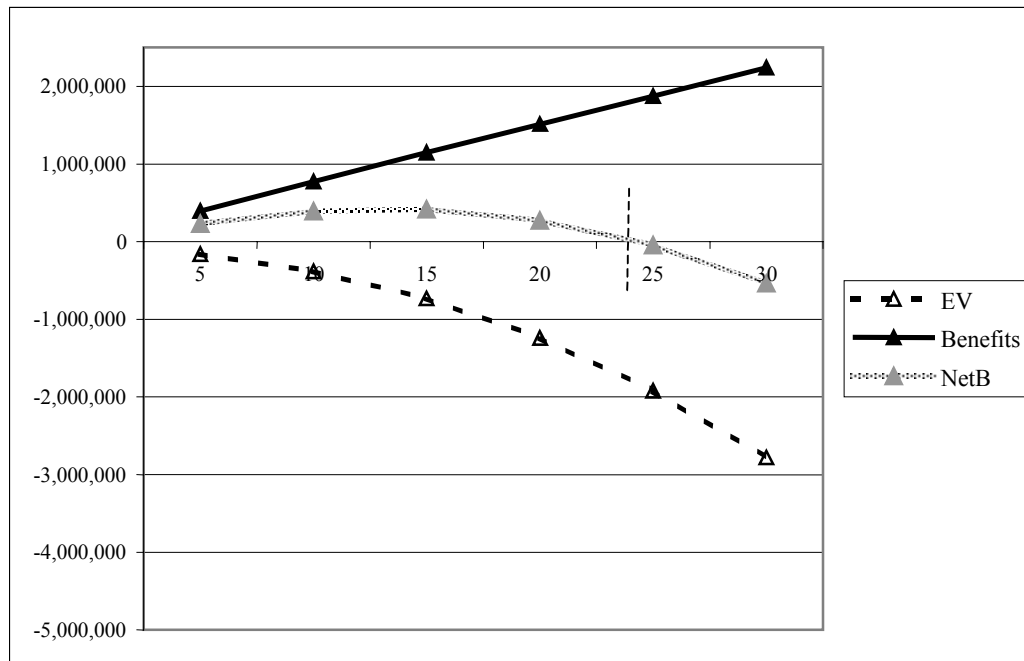


Table 5.3: Comparison of Mortality Benefits Estimates of CO₂ Reductions

<i>Study</i>	<i>Lives saved per MtC reduction</i>	<i>Scenario Assumptions</i>
Bussolo and O'Connor (2001)	334	India, 2010 15% CO ₂ reduction
Garbaccio <i>et al.</i> (2000)	298	China, 2010: 10% CO ₂ reduction
Dessus and O'Connor (1999)	100	Chile, 2010: 10% CO ₂ reduction
Cifuentes <i>et al.</i> (1999)	89	Chile, 2020: 13% CO ₂ reduction
Abt Associates (1997)	82	USA, 2010: 15% CO ₂ reduction

Source: O'Connor (2000); Bussolo and O'Connor (2001).

These health benefits of climate policy can also be expressed in value terms. Even if in India the number of premature deaths averted per tC abated is quadruple that in the United States, in value terms the difference will be smaller, given India's much lower *per capita* income, hence willingness (capacity) to pay for cleaner air. The ancillary benefits per tC abated in India come to around \$58 (at 1995 exchange rate). This compares with one early U.S. estimate of around \$26/tC from emission reductions in two sectors¹⁷ – transport and electricity – which together account for about two-thirds of carbon emissions (Ayres and Walter 1991). A more recent review for the U.S.A. by Burtraw and Toman (1997) reports on the results of eight studies whose mean estimate of ancillary benefits is virtually identical to the Ayres and Walter figure, with a low estimate of around \$3 and a high of \$89¹⁸.

European ancillary benefit estimates tend to be higher than those for the United States, reflecting in part higher population exposure (due to higher urban population densities and prevailing winds that blow pollution inland in Europe but out to sea from the eastern United States). Considering only emission-related benefits, the values range (in 1990 \$US) from a low of \$20/tC (Barker, 1993, for the U.K., based on social preferences revealed from the marginal costs of implementation of existing abatement technologies) to a high of \$212/tC (Alfsen *et al.* 1992, for Norway; Pearce 1992 reports a similar figure for the U.K.: \$195/tC). A mean value of emission-related benefits, based on the estimates reported in Ekins (1995), is around \$100/tC. A more recent Europe-wide assessment (Barker and Rosendahl, 2000) for the first Kyoto Protocol control period (2008-2012) finds average ancillary benefits per tC equivalent of around \$125 (at 2000 prices and exchange rate). These benefits amount to between 15 and 40 per cent of the change in GDP brought about by GHG mitigation measures, depending on the European country.

6. CONCLUDING OBSERVATIONS

This paper has laid out the theory, methodology and some empirical results from the use of economy-wide models to compare ancillary benefits to the primary costs of climate policy in developing countries. A few results bear repeating.

First, the top-down modelling approach depends on the question at hand: it is best suited to providing broad guidance to national policy makers who are concerned principally with the economic costs of a carbon constraint. By integrated ancillary benefits into a consistent modelling framework, it is possible to estimate the “no regrets” level of abatement effort.

Second, the hypothesis seems to be borne out that countries that have undertaken relatively little prior abatement of local air pollution stand to reap larger ancillary benefits of climate policy than countries that have. While this does not always break down along developing/developed country lines, the environmental Kuznets curve observed in some of the data on pollution levels *vs. per capita* income suggests this is probably a good first approximation.

Third, this still leaves open the question of whether a country could get more for its money, in terms of local pollution-related health benefits, by focusing on local air quality targets rather than on greenhouse gas reductions. Limited evidence suggests that there may be tradeoffs but that there can be considerable coincidence between cost-effective local pollution control measures and cost-effective GHG control measures. In any event, the intent of the analysis is not to suggest that policy makers ought to focus on climate policy first. Rather, it is to call their attention to (and attempt to quantify) previously overlooked or undervalued benefits of such policy, while fully recognising that their priorities may well lie with local air pollution. In any event, the question of the consistency of orderings between local abatement and global abatement cost-effectiveness is an area requiring more country-specific research.

Fourth, the evidence suggests that, while dose-response function transfer from developed to developing countries is justifiable for mortality from particulates exposure, it is more problematic where the endpoint being measured involves an explicit behavioural component, like self-medication, clinic visits, or hospital admissions. Even more problematic is transfer of *WTP/VSL* estimates, especially given the sensitivity of the derived value for the target site to the assumed income elasticity of *WTP*, on which empirical evidence to date is far from consistent.

Fifth, there is a sense of incompleteness in an analysis that considers only the ancillary (not the primary) benefits of climate policy – bearing in mind that costs are simply negative benefits and that some countries might gain on net from climate change. The rationale for this focus has been that developing country policy makers are likely to discount heavily the distant future and highly uncertain primary benefits, while the ancillary ones are far more certain and immediate. Still, in the final analysis, all significant costs and benefits will need to be compared, as will the timing of controls.

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TECHNICAL APPENDIX: ELEMENTS OF THE MODEL STRUCTURE

This Appendix presents the basic mathematical structure of the production and consumption sectors of the CGE models used in the analysis; also the determination of aggregate investment . (See Bussolo *et al.* 2001 for further details.)

Production structure

Recall text Figure 2.2 that presents the production nesting.

At the top level, the producer chooses a mix of a value-added-plus-energy¹⁹ aggregate (Q^{KEL}) and an intermediate demand aggregate (N^D). The optimisation problem takes the following form:

$$\min P_j^{KEL} Q_j^{KEL} + P_j^N N_j^D$$

subject to the production function:

$$XP_j = \left[a_j^{KEL} Q_j^{KEL \rho_{l,p}} + a_j^N N_j^{D \rho_{l,p}} \right]^{1/\rho_{l,p}}$$

where P_j^{KEL} is the aggregate price of value added plus energy, P_j^N is the price of the intermediate aggregate, a_j^{KEL} and a_j^N are the CES share parameters, and ρ is the CES exponent. The exponent and the CES elasticity are related via this relationship:

$$\sigma = \frac{1}{1-\rho} \Leftrightarrow \rho = \frac{\sigma-1}{\sigma}.$$

Note that in the model, the share parameters incorporate the substitution elasticity using the following relationships:

$$\alpha_j^{KEL} = (a_j^{KEL})^\sigma \quad \text{and} \quad \alpha_j^N = (a_j^N)^\sigma$$

Each subsequent production nest in Figure 2.2 is subject to an equivalent optimisation rule.

Consumption structure

Consumers under the ELES are assumed to maximise the following utility function:

$$\max U = \sum_i \mu_i \ln(C_i - \theta_i) + \mu_s \ln\left(\frac{S}{P}\right)$$

subject to the budget constraint:

$$\sum_i P_i^C C_i + S = Y^d$$

C is consumer spending, S is saving (in value), Y^d is disposable income, P^C are consumer prices, and μ and θ are the ELES parameters.²⁰ The Engel aggregation condition requires the following constraints on the parameters μ :

$$\sum_i \mu_i + \mu_s = 1$$

The following demand functions can be derived:

$$C_i = \theta_i + \frac{\mu_i}{P_i^C} \left(Y^d - \sum_j P_j^C \theta_j \right)$$

The usual interpretation of this demand function is that consumption is composed of two parts. The first part has been referred to as the subsistence minimum (or floor consumption), θ . The term in parenthesis represents residual income after subtracting expenditures on the subsistence minima. Therefore the second part of consumption of any given sector's output is a share of supernumerary income. Note that there is no minimal consumption of savings, i.e., θ_s is 0. Savings can be determined via the budget constraint:

$$S = Y^d - \sum_i P_i^C C_i$$

The income and price elasticities are given by the following formulae:

$$\eta_i = \frac{\mu_i Y^d}{P_i^C C_i} = \frac{\mu_i}{\chi_i}$$

$$\varepsilon_i = \frac{\theta_i (1 - \mu_i)}{C_i} - 1$$

The income elasticity is equal to the ratio of the marginal propensity to consume good i out of supernumerary income, μ , over the average propensity to consume good i out of income.

Investment determination

Appendix Table 1 includes the equations for the closure of the saving and investment account. The domestic India-wide *value* of investment – the product of investment price index P^I , and the investment volume I^{TOT} – is equal to domestic saving (households and government savings) plus foreign saving, plus depreciation, less expenditure on building stock.

Appendix Table 1: Determination of Aggregate Investment (Exogenous Foreign Saving)

$$P^I I^{TOT} = S_h^{TOT} + S_g + P^{SAVF} S_f + Y^{DEPR} - \sum_i P A_i X_i^{ASTOCK}$$

$$P_r^I I_r^{TOT} = \alpha_r P^I I^{TOT}$$

$$S_f = P^{SAVF} \bar{S}_f$$

NOTES

¹ Pearce (2001) cites damage estimates for non-OECD countries from a doubling of atmospheric CO₂ equivalent concentration for non-OECD countries of between 1.6% of GNP (Fankhauser 1995) and 2.7% of GNP (Tol 1995).

² Biomass fuel is normally a by-product of another productive sector – whether agriculture, livestock, or forestry. While the marketed outputs of these sectors get reflected in the I/O tables, it is not legitimate to apply the same technical coefficients to these by-products.

³ Dowlatabadi (1998) provides some illustrations of how endogenising technical change can affect the analysis of optimal climate policy. For example, with induced technical change, postponing climate policy significantly raises costs of achieving the agreed target by foregoing important learning opportunities. Goulder and Schneider (1999) extend this analysis, suggesting that (in the absence of prior R&D tax distortions) induced technical change can increase the gross social costs of a given carbon tax, though this is more than offset by increased benefits. Thus, the optimal level of abatement is higher. Viewed differently, for a given abatement target, the gross social costs are lower with induced technical change than without.

⁴ VSL can be alternately expressed as the value of a premature death avoided and is estimated by individual willingness to pay (WTP) for small reductions in the risk of premature death from specific causes (whether on-the-job accidents or pollution-related illness). Algebraically, $VSL = \text{Value of Arisk} / \text{Arisk}$. So, for instance, if the average WTP for a 1/100,000 reduction in the risk of premature death (say, associated with a 10 μm^3 reduction in PM₁₀) is \$50, then the VSL implied by that WTP is \$50/(1/100,000), or \$5 million.

⁵ To illustrate the role of temperature and humidity, according to one estimate, concentrations of particulate matter from a fixed quantity of emissions in hot and dry regions are about one-third of what would be expected from the same emissions under most other climatic conditions (Working Group 1997).

⁶ Ideally, region- or city-specific information on atmospheric conditions can be found to determine these frequencies, but if not then certain “default” frequencies can be used as an approximation.

⁷ Particle size is measured in terms of “aerodynamic diameter” and is given in units of microns (thousands of a millimeter); thus PM₁₀ refers to particles with diameter of 10 microns or less and PM_{2.5} has an equivalent interpretation.

⁸ The relationship between PM₁₀ and mortality is non-linear, however, with a change evaluated at 50 $\mu\text{g}/\text{m}^3$ associated with a 1.4 per cent increase in mortality (i.e., closer to the U.S. means) and one evaluated at 150 $\mu\text{g}/\text{m}^3$ increasing mortality by 0.4 per cent.

⁹ Likewise, a recent study for Santiago, Chile, revises down earlier estimates by Ostro of the particulates dose-response relationship. While Cifuentes *et al.* (2000) broadly confirm the statistical significance of the relationship found by Ostro *et al.* (1996) for PM_{2.5}–mortality, the dose-response coefficient is found to be only about half as large, with the difference attributable to controls for seasonal effects.

¹⁰ In the case of the United States and India, as of 1998 life expectancy at birth was 77 and 63 years respectively; WDI Indicators CD-ROM of the World Bank.

¹¹ This raises the vexed question of whether a sample of Canadians is representative of anyone other than, well, a sample of Canadians ;).

¹² It is too early to present results of policy simulations for China, so the discussion in section 5 focuses just on health impacts.

¹³ IEA (1995), for example, estimates India’s 1992 energy use per tonne crude steel at 0.94 toe, compared to 0.34 in the Republic of Korea and 0.33 in Japan (Table 5.10).

¹⁴ Roy *et al.* (1999) suggest, e.g., that while raising energy prices in India would be an effective carbon abatement policy, it could be costly given what they estimate to be relatively weak inter-input substitution possibilities. If a policy maker shared this assessment, then s/he might be interested in knowing the effect of assuming lower substitution elasticities than the central values used in Bussolo and O’Connor (2001).

¹⁵ See, e.g., Brandon and Hommann (1995); Simon *et al.* (1999).

¹⁶ See Burniaux (2000) for a multi-gas assessment of the Kyoto Protocol, in which the savings from abating across three gases rather than just CO₂ averages \$26/tCeq for the Annex I countries in 2010.

¹⁷ The benefits estimates are based on the assumption of a 20% reduction in air pollution from 1978 levels.

¹⁸ Burtraw and Toman propose a “rule-of-thumb” (for the United States) that ancillary benefits can be assumed to be roughly 30% of the cost of carbon reduction for low to moderate rates of abatement. Observing that over some range the marginal costs of GHG reductions are likely to be close to zero, they

conclude that the existence of ancillary benefits even as small as \$3/tC could significantly increase the volume of emissions reduction that is considered “no regrets” in the sense of having negative or zero net cost.

¹⁹ Due to the crucial importance of energy in terms of pollution, the demand for energy has been separated from the rest of intermediate demand, and incorporated in the value added nest.

²⁰ In the utility function, S needs to be deflated by an appropriate price, which would represent the consumer spot price of future consumption. This price does not need to be specified for the model since household saving can be derived as a residual from the budget constraint. For welfare calculations, the consumer price index, cpi , has been chosen as the saving deflator since there is no forward-looking behaviour in REGEMI.